TECHNICAL BACKGROUND DOCUMENT FOR THE SUPPLEMENTAL REPORT TO CONGRESS ON REMAINING FOSSIL FUEL COMBUSTION WASTES

GROUND-WATER PATHWAY HUMAN HEALTH RISK ASSESSMENT

Revised Draft Final

June 1998

U.S. Environmental Protection Agency
Office of Solid Waste
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Washington, DC 20460

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1. INTRODUCTION

Section 3001(b)(3)(C) of the Resource Conservation and Recovery Act (RCRA) exempts fossil fuel combustion (FFC) wastes from regulation as hazardous wastes pending completion by the U.S. Environmental Protection Agency (EPA) of a report to Congress detailing the Agency's findings regarding the need for more strict regulation of these wastes. Specifically, Section 8002(n) of RCRA directs EPA to prepare a report examining the following:

- Source and volumes of FCC wastes generated per year
- Present disposal and utilization practices
- Potential danger, if any, to human health and the environment from the disposal and reuse of such material
- Documented cases in which danger to human health or the environment from surface runoff or leachate has been proved
- Alternatives to current disposal methods
- Costs of such alternatives
- Impacts of those alternatives on the use of coal and other natural resources
- Current and potential utilization of such materials.

EPA completed its *Report to Congress on Wastes from the Combustion of Coal by Electric Utility Power Plants* (EPA 1988) in 1988, but failed to complete a regulatory determination of FFC wastes at that time. In 1992, the Bull Run Coalition (an Oregon citizens group) and the Edison Electric Institute, acting as intervenors, sued the Agency to complete a regulatory determination for FFC wastes. Pursuant to the suit, EPA entered into a consent agreement that established the schedule according to which EPA would complete its decision-making activities relating to FFC wastes. EPA agreed to complete a regulatory determination for fly ash, bottom ash, boiler slag, and flue gas desulfurization wastes from coal-fired electric utilities by August 1993 and deferred its decision on all remaining fossil fuel combustion wastes until April 1998, pending completion of additional study.

In 1993, EPA issued a regulatory determination exempting from hazardous waste regulation fly ash, bottom ash, boiler slag, and flue gas desulfurization (FGD) wastes generated by coal-fired electric utilities and independent power producers (IPPs) when such wastes are managed alone (58 *FR* 42466, August 9, 1993). The determination, however, explicitly excluded certain categories of FFC wastes from the permanent exemption pending additional study. These so-called remaining wastes included wastes from coal-fired electric utilities that are comanaged with low-volume wastes, wastes from the combustion of other fossil fuels, wastes from fluidized bed combustion, and FFC wastes from non-utilities.

EPA is preparing the supplemental report to Congress on remaining FFC wastes (pending completion). As part of this study, EPA examined the potential danger to human health and the environment from remaining FFC wastes arising from predominant disposal and beneficial use practices. This report presents the methodology and results of EPA's assessment of human health risks resulting from ground-water contamination from remaining FFC waste management.

1.1 PURPOSE OF STUDY

The primary objective of this study is to determine the potential for harm to human health and the environment resulting from remaining FCC waste management practices to support the Remaining FFC Waste Supplemental Report to Congress. Previous work by EPA concluded that the greatest potential for harm from FFC wastes was associated with the potential for ground-water contamination (EPA 1988, EPA 1993). Current EPA risk assessment policy also encourages consideration of comprehensive human health risk and ecological impacts (EPA 1995a, EPA 1995b). Accordingly, the remaining FFC waste risk assessment included two components: the ground-water pathway human health risk assessment and the above-ground multi-pathway human health and ecological risk assessment. This report presents the technical approach to and results of the ground-water pathway human health risk assessment. The results of the above-ground multi-pathway human health risk assessment, which was conducted in close coordination with the ground-water study, are presented under separate cover (EPA/RTI 1998).

Because the remaining waste universe is both large and diverse, representing thousands of facilities spread throughout all 50 states, EPA could not gather sufficient site-specific data to completely characterize the actual and potential damages to ground water and human health.[1] Accordingly, EPA attempted to assess nationwide ground-water pathway risks, as realistically as possible, through quantitative modeling. To do so, EPA developed representative waste management scenarios that reflect the variability of waste characteristics, waste management practices, and management unit environmental settings observed or suspected to occur at sites throughout the nation. Using EPA's *Composite Model for leachate migration with Transformation Products v1.2* (EPACMTP), EPA's ground-water fate and transport model, EPA assessed the individual human health risk resulting from exposure to ground water contaminated from each of the waste management scenarios. A detailed description of EPACMTP is presented in the *EPACMTP User's Guide* (EPA 1995c).

1.2 SCOPE OF REPORT

As discussed in detail below, EPA considered the risks from four remaining FFC waste categories independently. These were comanaged wastes from coal-fired electric utilities[2], wastes from oil-fired utilities, wastes from fluidized bed combustion (FBC), and wastes from coal-fired non-utilities. For each of these categories, EPA studied the waste management practices commonly employed throughout the relevant population of waste generators and suspected to present the greatest potential for release to the environment. For all four remaining waste categories, for example, EPA examined the risks from management in unlined landfills. Further, EPA examined the risks from mine placement of FBC and coal-fired utility comanaged wastes. EPA studied the risks from unlined surface impoundment management of coal-fired utility comanaged wastes and oil ash. In addition, EPA studied the risks associated with the management of non-utility wastes and oil-fired utility wastes in unlined commercial landfills receiving other industrial wastes.

[1] The results of EPA's assessment of actual damages to human health and the environment, while germane to this study, are presented in Chapter 5 of the Supplemental Report to Congress, and are only referenced in this report.

[2] Comanaged wastes are large-volume combustion wastes (i.e., fly ash, bottom ash, boiler slag, and/or flue gas desulfurization sludge) that are mixed with low-volume wastes (e.g., pyrites, boiler cleaning points). Section 3 discusses comanagement in detail.

For each waste and management scenario examined, EPA considered the risk to an individual adult receptor from exposure to ground water contaminated by the release of leachate from the waste management unit. EPA also explored two methods for considering the potential risk to an individual child resident (presented in Section 6). After preliminary analyses of a range of constituent groups, EPA limited its focus to the risks associated with metals contamination. The metals studied were limited to those for which (1) waste characterization data were available, (2) current toxicological information was available to serve as a basis for calculating a health-based risk benchmark value or for which another suitable benchmark value could be obtained, and (3) metals adsorption behavior had been sufficiently characterized to support modeling with EPACMTP.

EPA did not consider mine placement of oil ash, because the Agency did not find evidence that any operator employs the practice. Similarly, EPA did not consider management of FBC wastes or non-utility wastes in surface impoundments because these practices were determined to be very rare. EPA also did not consider mine placement of non-utility remaining wastes because the risks from any such projects were thought to have been adequately captured by the larger scenarios involving FBC and comanaged wastes. With respect to beneficial uses, EPA limited its review to the practices in which remaining wastes were found to be placed directly on the ground (e.g., minefills and agricultural applications[3]) and did not consider uses that would result in extensive modification of the wastes (e.g., vanadium recovery from oil ash) or require incorporation of the waste into products (e.g., FGD sludge in wallboard manufacture, ash incorporation into cement and aggregate).

In addition, EPA did not consider several minor categories of remaining FFC wastes, including other fossil fuels (e.g., petroleum coke, Orimulsion®), wastes from emerging combustion technologies (e.g., pressurized fluidized bed combustion), and wastes from co-burning of fossil fuels and other materials (e.g., tires, solvents, other wastes). Because of insufficient data on waste characteristics and volumes and/or waste management practices, these categories could not be studied.

[3] Agricultural application was considered in detail in the above-ground risk assessment. EPA did not model this use for ground-water risks because the associated contaminant loading was expected to be very small compared with the loading associated with unlined landfills.

1.3 ORGANIZATION OF REPORT

The remainder of this report comprises six sections:

- Section 2 discusses in detail the methodology of the risk assessment.
- Section 3 profiles the remaining waste universe by describing the industry characteristics, waste quantities and characteristics, and waste management practices for each of the four remaining waste categories.
- Section 4 describes each of the remaining waste scenarios that were modeled.
- Section 5 presents the results obtained for each of the scenarios.
- Section 6 briefly discusses the risks to children exposed to waters contaminated from remaining wastes.
- Section 7 discusses the uncertainties present in EPA's data and methods and the potential influence they may have on result interpretation.
- Section 8 presents a summary and conclusions.

In addition, the report includes numerous technical appendices.

2. METHODOLOGY

The remaining waste universe includes four sectors: coal-fired utility comanaged wastes, oil ash, fluidized bed combustion (FBC) wastes, and non-utility fossil fuel combustion (FFC) wastes. EPA found that each sector represented a distinct generator population with chemically distinct byproducts. Moreover, each remaining waste category varied in waste stream characteristics, waste management practices, and the geographic distribution of waste management facilities. For example, in all remaining waste categories, concentrations of some metals observed in waste leachates were found to vary by up to three or four orders of magnitude. Similarly, coal-fired utility comanaged waste units varied in size from as small as a few acres to more than 1,500 acres, and operators reported comanaging from 1 to 15 different low-volume waste streams along with 1, 2, or 3 different large-volume wastes. Comanaged waste sites, FBC sites, and non-utilities were found throughout the entire United States, while oil-fired sites were found predominantly in the Northeast and Southeast.

Given the wide diversity of the remaining waste universe, EPA elected to study the risks from each remaining waste generator category separately. To do so, EPA collected available data describing waste characteristics, waste management practices, and, where possible, environmental setting and performance, for each of the four sectors. The data assembled for each sector included site-specific and nationwide waste, management, hydrogeology, and meteorology data. These data were compiled into sector-specific databases, from which representative waste profiles and waste management scenarios could be derived. Each of the generator categories was subjected to a multistep assessment process. This process includes screening, deterministic modeling, and probabilistic modeling to determine the nationwide potential for risks to human health from ground-water contamination.

For all of the sectors, EPA faced significant challenges relating to the availability and representativeness of data. EPA addressed these challenges by incorporating appropriate steps into the study design. For example, EPA used the high-end (95th percentile) value of waste characteristics to overcome the generally low availability of waste characterization data relative to the potential variability of characteristics and waste combinations. Similarly, EPA performed focused

sensitivity analyses to determine those parameters most able to influence model output. By setting these variables at their respective high-end values, EPA ensured a very conservative estimate of risk. Finally, EPA performed probabilistic analyses of each model scenario to capture the effects of parameter variability on potential risk and to demonstrate that the deterministic model results were conservative.

The following sections introduce each step of the risk assessment methodology and the driving assumptions made for each remaining waste category. Later sections of the report present more details on specific issues.

2.1 CONSTITUENTS OF CONCERN

EPA began with a thorough review of existing information to determine what constituents appear in the remaining FFC wastes. EPA initially considered four categories of constituents: heavy metals, other inorganics, conventional organics, polychlorinated diebenzofurans and polychlorinated dibenzodioxins (PCDDs and PCDFs), and radionuclides. On the basis of these reviews, EPA eliminated all but the first category as constituents of concern for remaining FFC wastes.[1]

For screening purposes, EPA considered all constituents for which waste leachate data were available and for which ingestion toxicity estimates or a maximum contaminant level (MCL) or action level (AL) (e.g., lead and copper) had been developed. Certain data required to run EPACMTP effectively, however, have not been developed for all of these constituents. Specifically, the adsorption isotherms that describe the tendency of a metal to remain bound to particle surfaces in ground water or enter the aqueous phase have been developed for only 16 metal species. Accordingly, in high-end and probalistic modeling, EPA considered only a subset of all constituents for which screening was performed. Table 2-1 depicts the metals considered.

[1] See Appendix L for a summary of findings regarding dioxins in FFC wastes, Appendix M for a summary of findings regarding conventional organics in FFC wastes, and Appendix N for a review of radionuclides in FFC wastes.

Table 2-1. Constituents Considered in Ground-Water Pathway Risk Assessment

Screening	Modeling
Antimony, Arsenic, Barium, Beryllium, Boron, Cadmium, Chromium III, Chromium VI, Copper, Flouride, Lead, Managanese, Mercury, Molybdenum, Nickel, Nitrate, Selenium, Silver, Strontium, Thallium, Vanadium, Zinc	Antimony, Arsenic, Barium, Beryllium, Cadmium, Chromium III, Chromium IV, Copper, Lead, Mercury, Nickel, Selenium, Silver, Thallium, Vanadium, Zinc

2.2 DEFINING THE PATHWAY RECEPTORS AND BENCHMARKS

An early step in the risk assessment process was defining the pathway receptor and the corresponding benchmark values. EPA assumed the primary receptor to be a nearby adult resident of an FFC waste management unit. The resident was assumed to drink tap water derived from ground water that had been affected by the waste management unit. EPA used accepted toxicity values and exposure assumptions to develop benchmark values reflecting the drinking water concentration of constituents of concern that would result in the target level risk.[2] Specifically, assumptions about the resident (how much s/he drinks, how often and for how long, his or her body weight) and about the constituent of concern (its toxicity) were combined into a single benchmark value for each constituent of concern. EPA derived a similar set of benchmarks for child receptors as discussed in Section 6. (Appendix B presents the details of fixed exposure assumptions, toxicity values, and resulting benchmark values.)

The calculated adult receptor benchmark values were used in the three assessment steps—screening, deterministic modeling, and probabilistic modeling. In screening, waste leachate concentrations were compared with the benchmarks directly. In deterministic and probabilistic modeling, the predicted concentrations of concern in ground water were compared with the receptor benchmarks.

2.3 SCREENING ANALYSIS

As described above, EPA began to assess the risks from remaining FFC wastes by comparing the concentrations of chemicals measured in FFC wastes directly to the calculated benchmark values,

[2] EPA selected a target risk level of 1×10^{-6} for carcinogens and a hazard quotient (HQ) of 1 for non-carcinogens.

or "risk-screening criteria." The purpose of the exercise was to determine the constituents of the waste leachate that could be expected to present no significant risk even when undiluted, and that therefore could be eliminated from further consideration. Waste constituents exceeding the screening criteria were considered further in subsequent assessment steps.

Typically, results of laboratory leaching studies, such as the Extraction Procedure (EP) [SW-846 Method 1310] or Toxicity Characteristic Leaching Procedure (TCLP) [SW-846 Method 1311] have been used in screening assessments to represent the composition of leachate escaping from solid waste disposal units. For oil ash and FBC wastes, EPA used EP and TCLP results in the following manner. For oil ash, EPA compiled the results of leachate analyses for over 86 samples of as-managed oil-fired utility wastes collected from over 30 plants[3] and identified the 95th rank-ordered percentile concentration[4] of each constituent of concern to compare with the corresponding benchmark values. Similarly, EPA compiled the results of 50 FBC waste leachate samples from over 30 sites and identified the 95th percentile value concentration of each constituent for screening.[5] By using the 95th percentile concentration, EPA ensured a conservative estimate of the expected leachate concentrations.

For utility comanaged wastes, EPA did not use EP or TCLP values in screening. Rather, EPA based its study on as-managed waste leachate samples collected *in situ* from various points within and beneath 18 sites with active or recently closed waste management units.[6] These samples were collected during an industry-sponsored study of the environmental performance of comanaged waste disposal units, and provided the best measure available to date of the variability of the waste

[3] These samples included settling basin solids, fly ash, and bottom ash. All of the oil ash characterization data are summarized in Appendix F.

[4] The 95th rank-ordered percentile is the concentration value below which 95 percent or more of all concentration values fell.

[5] These samples included fly ash, bed ash, and combined ash. All of the FBC waste characterization data are summarized in Appendix F.

[6] These samples included porewater samples from drill cores from impoundments and landfills, reflecting fly ash, bottom ash, and/or FGD sludge in combination with various low-volume wastes. All of the comanaged waste characterization data are summarized in Appendix F.

characteristics owing to waste combinations, waste management practices, region and climate, and/or other factors. EPA did not identify sufficient non-utility waste characterization data to develop a representative profile of these wastes. Instead, EPA assumed that utility comanaged wastes provided a reasonable approximation of the waste characteristics of non-utility wastes.

Unlike previous special waste studies, this screening analysis did not apply a dilution and attenuation factor (DAF).[7] Preliminary modeling indicated that for some FFC waste scenarios, the predicted DAF may be less than 10. Accordingly, application of such a factor would have resulted in the possibly erroneous conclusion that a constituent posed negligible risk. The selection of DAF=1 implies that screening reflects human exposure to raw, undiluted leachate, a very conservative assumption. Consequently, most constituents observed in waste management unit leachate were retained for further evaluation. Section 5 presents the results of the screening analysis.

2.4 DETERMINISTIC MODELING (HIGH-END) ANALYSIS

Those metals found to exceed benchmark values in the screening assessment were evaluated further through deterministic fate and transport modeling. The deterministic high-end assessment used EPACMTP to predict the extent of ground-water contamination that might result from each representative management scenario for all constituents of concern. Each management scenario was described in terms of waste characteristics, unit size, unit liner characteristics and infiltration rate, unit meteorological and hydrogeological setting, and proximity to the nearest ground-water receptor well. EPACMTP then calculated the movement of leachate escaping the unit over a study period of 10,000 years, and calculated a peak concentration observed for the nearest receptor well. EPA calculated the predicted risk by comparing the peak down-gradient concentration with the benchmark value.[8]

[7] Cement Kiln Dust (EPA 1993b) and Mineral Processing Waste (EPA 1990) studies each employed a DAF of 10 in performing the risk screening step. Similarly, EPA employed a DAF of 10 in considering ground-water pathway risks in its 1993 regulatory determination on FFC wastes. However, 90th percentile values for ingestion and exposure duration were also used in each of the previous studies, roughly canceling the effect of the DAF.

[8] The ratio of the predicted concentration to the benchmark yields an HQ for non-carcinogens, and a risk value for carcinogens. An HQ> 1 indicates a potential risk, and risk value greater than 1×10^{-6} indicates a potential risk.

The high-end deterministic assessment was designed to provide an estimate of risk that is greater than or equal to the risk expected from any actual site. To establish the high-end scenarios, EPA determined a median and 95th percentile value for all model input parameters for each FFC waste generator category. EPA then performed a targeted sensitivity analysis to identify those parameters with the greatest influence over model results. Finally, the two most sensitive parameters (waste concentration and receptor well location) were set at their respective high-end values while all other parameters were set at median values. (See Appendix K for a detailed discussion of the sensitivity analyses.)

EPA employed a wide range of data sources to develop appropriate distributions for each of the model parameters for each of the waste management scenarios. Appendix A lists and briefly discusses the input values selected for each scenario. Other appendices discuss specific parameters in greater detail. For example, Appendix F reviews in detail all waste characterization information considered in the risk modeling exercises.

EPA used site-specific information wherever possible. For example, the Electric Power Research Institute (EPRI) provided detailed waste management unit dimensions for oil-fired utility solids settling basins (EPRI 1998). These statistics were used to develop a distribution of units from which unit characteristics were developed for one of the oil-specific model scenarios. In the absence of site-specific data, EPA searched for more broad industry-specific information. For example, EPA's 1990 National Interim Particulate Inventory (US90) database provides location, capacity, and fuel usage information for the largest non-utility coal-fired boilers in the nation. EPA used these data, coupled with other information sources, to develop geographic, climatologic, and waste generation distributions for the non-utility scenarios.

Where necessary, EPA relied upon data distributions assembled for other purposes but believed to be suitable for the FFC risk assessment. Most importantly, EPA relied upon or extrapolated from the Hazardous Waste Identification Rule (HWIR) data sets presented in the 1995 EPACMTP Users Guide (EPA 1995c) for some aquifer characterization and hydrogeological data (see Appendix E). For example, EPA did not identify sufficient site-specific information to develop a representative distribution of depth of the unsaturated zone underlying oil-fired utility sites. In this

case, EPA selected from the HWIR data set only those data appropriate to the states in which oil-fired utilities operate to develop a scenario-specific distribution of depth to ground water. Section 4 describes in detail parameter selection and assumptions.

Once the high-end scenarios were defined, EPA ran each model for all constituents of concern that had survived the screening analysis. The output of each model run was the concentration of a single constituent predicted to occur in a nearby well used for drinking water. As with the screening assessment, EPA compared this concentration directly with the calculated benchmark values to determine the predicted risk. Section 5 presents the results of the high-end analysis.

2.5 CENTRAL TENDENCY (MONTE CARLO) ANALYSIS

In addition to the deterministic high-end analysis, EPA used the Monte Carlo capabilities of EPACMTP to perform a probabilistic assessment of risk for each waste management scenario. The purpose of the Monte Carlo analysis was to examine the distribution of risk that resulted when each of the model's input parameters was allowed to vary independently. EPA performed a Monte Carlo simulation for each scenario for each metal that was modeled in the high-end analysis. Each Monte Carlo simulation involved 2,000 iterations. The output of these iterations was used to develop a scenario- and constituent-specific distribution of risk results. To determine the relative conservatism of the high-end assessment, EPA compared the results of the high-end analysis with the Monte Carlo distribution.

2.6 MASS BALANCE AND COORDINATION WITH ABOVE-GROUND MODELING

As stated previously, EPA considered the risks to human health and the environment in two distinct studies: a below-ground study that is the subject of this report to assess the risks to humans resulting from exposure to contaminants in ground water contaminated by fossil fuel combustion wastes, and an above-ground study to assess the risk to humans and the environment from direct and indirect exposure to contaminants in above-ground media (e.g., soils, plants, air). EPA made every effort to coordinate the above-ground and below-ground studies. For example, each study considers

the same set of waste management practices exemplified by the same waste management scenarios[9] and described as having the same unit characteristics (e.g., area, height, capacity, project duration, and waste characteristics). This section discusses some of the coordination issues.

2.6.1 Mass Balance

Material removed from a landfill by water or wind erosion is no longer available for leaching. Accordingly, EPA considered the potential impact of the above-ground model in reducing the potential leachate generation or concentration in the ground-water model. As one instance of this, EPA assumed that the above-ground model allowed erosion to remove a certain amount of waste from the landfill in each year. EPA found that the total mass of material that would be eroded from the landfill was trivial with respect to the total quantity of material contained in the landfill. Accordingly, EPA concluded that wind and water erosion could not diminish the mass of the landfill sufficiently to affect the total contaminant flux via infiltration. As a result, the ground-water model assumed that waste in the landfill was not reduced by erosion, and thus there would be no effect on the leachate.

Likewise, EPA considered the potential for leachate to affect ground water at a site where wind-eroded wastes would be deposited. However, using the same reasoning as above, the total flux of contaminants from the eroded materials must be small compared with the flux from infiltration from the landfill mass and so did not model leachate from a deposition site.

EPA also considered the potential for leaching processes to reduce the concentration of available metals in waste that would be eroded from the landfill, thereby causing the above-ground model to overstate risks. EPA dismissed this concern as well. Because intermediate cover was assumed to cover previous years' wastes, the above-ground landfill was assumed to erode only recently emplaced wastes. Accordingly, only a short period of weathering would have operated on

[9] The one exception is land application. The dimensions of the land application scenario are such that any contribution of these projects to ground-water contamination would be small compared with the contributions from the landfill scenario, and the risks would thus be bounded by the landfill scenario. Therefore, EPA did not model ground-water pathway risks for the land application scenario.

the waste that could be eroded in the model. Relatively little of the available metals would have been removed via leaching in that short period, so the above-ground model did not account for this reduction.

2.6.2 Receptors

The ground-water pathway human health risk assessment determined the potential risks to nearby adult and child residents exposed to contaminated groundwater via drinking water ingestion. EPA determined that such general residents were the most appropriate receptors for the ground-water pathway. The above-ground assessment considered several more precisely defined receptors. For example, the above-ground pathway considered a subsistence farmer living adjacent to a waste management unit. The subsistence farmer was exposed to FFC waste constituents via direct inhalation and incidental ingestion, as well as by indirect ingestion through contaminated foodstuffs. It would have been possible to expose the subsistence farmer to ground-water borne contaminants also. However, some of the standard exposure assumptions are very different for the adult resident of the ground-water study and the subsistence farmer of the above-ground study. For example, the exposure duration and the body weight assumptions do not match for the two receptors. Accordingly, the risks can not be added between the two pathways directly.

2.6.3 Location

The relevant phenomena influencing the release, fate, and transport of waste constituents did not overlap between the above-ground and ground-water pathways, with the exception of certain meteorological conditions. For example, wind erosion and transport contributed the greatest portion of offsite deposition in the above-ground assessment, whereas infiltration and leaching to the subsurface was the only release contemplated in the ground-water pathway. In each study, EPA sought to identify median and high-end locations presenting conditions favoring the relevant release mechanisms. The locations favoring wind erosion were not those favoring leaching. As a result, EPA did not attempt to identify a single location that concurrently presented the worst case conditions with regard to all release pathways modeled. Further, since the conditions promoting the risks predicted for one pathway do not correspond to the conditions that would result in the risks predicted by another pathway, so the risks from the two studies cannot be added.

2.6.4 Period of Activity Examined

The above-ground study considered the potential for wind erosion and transport of wastes from active landfills and closed impoundments that have been dewatered and have received no cover material. In contrast, the ground-water pathway impoundment scenario assumed that wastes would be removed from the impoundment at the end of the active lifetime of operations. Moreover, the landfill scenario assumed that no leaching occurred during the active lifetime and began only at the end of operations and closure. Therefore, the two scenarios (appropriately) examined the potential risks from landfills and surface impoundments at different points in the project lifecycles.

EPA is aware that many comanaged waste surface impoundments represent the final resting place of the wastes. EPA believes, however, that the post-operational period of impoundments is captured adequately by the landfill scenario, which begins leaching at landfill closure. Moreover, EPA believes that the only relevant period of activity for the windblown transport at a landfill that is covered at closure is during waste placement. Therefore, the fact that the two studies did not examine the risks arising during the same period of activity of the landfills and impoundments should not affect the veracity of the results.

2.7 INTERNAL CONSISTENCY OF MODELING PARAMETERS

A common criticism of Monte Carlo analyses has been that physically meaningless combinations of input parameters may be selected by the model and may even drive model results in extreme cases. A similar criticism could be leveled at deterministic analyses in which the median value of covariant parameters is calculated independently, resulting in an improbable or impossible combinations of values.[10] EPA attempted to develop realistic and internally self-consistent deterministic scenarios by validating, through comparison with HWIR distributions, that combinations of potentially covariant parameters fell within reasonable ranges. Similarly, EPA enhanced the internal consistency of modeling parameters in the Monte Carlo analyses by linking critical parameters (infiltration and recharge rate, depth to and depth of the underlying aquifer, hydraulic gradient, and

[10] An extreme example of these modeling errors includes matching a site's infiltration rate of 2 meters per year with a depth to the uppermost aquifer of 30 meters and a horizontal hydraulic gradient near 0.

aquifer hydraulic conductivity) for each site considered in the analysis and prohibiting independent selection of those parameter values.

3. PROFILE OF REMAINING WASTES

This section profiles the remaining fossil fuel combustion (FFC) waste universe and includes background on the industries, wastes, and waste management practices studied in this report. A more comprehensive discussion of these topics is available in the supplemental report to Congress. The following subsections describe each of the four remaining FFC waste categories:

- Coal-fired utility comanaged wastes
- Oil-fired utility wastes
- Fluidized bed combustion (FBC) wastes
- Non-utility combustion wastes.

For each category, the relevant subsection discusses the relative significance of the waste-generating industry, outlines the combustion technologies used, and presents the geographic distribution of combustion facilities. Each subsection then provides a brief overview of waste generation rates and waste characteristics and concludes with a discussion of predominant waste management practices.

3.1 COAL-FIRED UTILITY COMANAGED WASTES

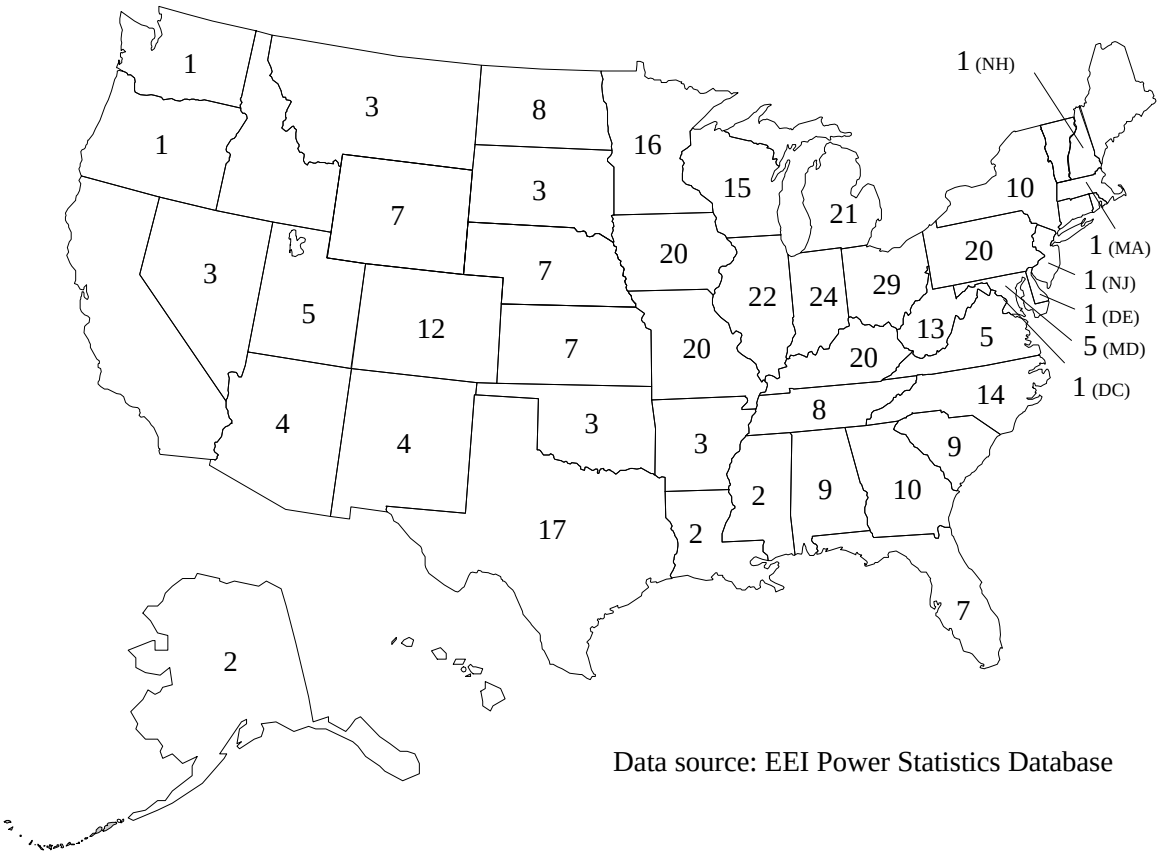
3.1.1 Industry

Coal is the primary fossil fuel used by electric utilities in the United States. In 1996, coal-fired utilities generated 1,737 billion kilowatt-hours of electricity, which represented 56.5 percent of all utility electricity generation nationwide (DOE 1998a). The production of energy from coal involves combustion of the fuel in a boiler to heat water and produce steam, which is then used to drive electricity-generating turbines. Wastes resulting from this combustion process include fly ash, bottom ash, boiler slag, and, for some facilities, flue gas desulfurization (FGD) wastes. Additionally, utilities generate a variety of liquids and solid wastes, which are collectively called low-volume wastes, from activities associated with coal combustion. Combustion wastes, when comanaged with other wastes, are a subject of this risk assessment.

The most common coal combustion technology used by utilities is the pulverized coal (PC) fired boiler. PC-fired boilers accounted for 92 percent of coal-fired generating capacity in 1994 (EEI 1994). PC-fired boilers burn finely ground coal in suspension at high temperatures. Under these conditions, coal burns very efficiently and completely, making the technology well suited to large capacity applications, including utility electricity generation. Utilities also use stokers and cyclones. Stokers burn coarsely crushed coal that is mechanically fed onto a grate inside a furnace. Cyclones are a specialized design used for burning low ash-fusion temperature coals. Stokers make up less than 1 percent and cyclones make up 8 percent of the coal-fired utility capacity. All three technologies employ similar supporting technologies for fuel storage and processing, steam generation, cooling, and equipment cleaning (Babcock & Wilcox 1992).

As shown in Figure 3-1, coal-fired power plants are distributed throughout the United States, with the largest concentrations in the Northeast and Midwest.

Figure 3-1. Location of Utility Coal-Fired Power Plants



Data source: EEI Power Statistics Database

3.1.2 Waste Volumes and Characteristics

Utilities generated almost 89 million tons of coal combustion wastes in 1994 (ACAA 1996). This total includes fly ash, bottom ash, boiler slag, and FGD sludge. More than half (62 percent) of the total is fly ash. The large proportion of fly ash is due to the predominance of PC-fired boilers, which generate mostly fly ash as a result of suspension firing, and the prevalence of high efficiency particulate collection in the utility sector. In addition to the four large-volume combustion wastes, supporting processes at coal-fired utilities generate significant quantities of low-volume wastes, which include the following:

- Boiler fireside washwater (e.g., air heater and precipitator washwater)
- Boiler chemical cleaning waste
- Boiler blowdown
- Cooling tower blowdown
- Coal pile runoff
- Coal mill rejects/pyrites
- Demineralizer regenerant and resins
- Waste from floor and yard drains and sumps
- Laboratory wastes
- Wastewater treatment sludge
- Water treatment sludge.

No comprehensive data exist on the total quantity of these wastes generated. For purposes of this report, however, the total quantity generated is less significant than the quantity comanaged. Section 3.1.3 presents information on the quantities of waste comanaged and the frequency with which comanagement occurs.

Many of the metals studied in this assessment can be detected in leachate from comanaged wastes. Porewater characterization data for comanaged coal combustion wastes in impoundments and landfills are presented in Table 3-1. These characterization data were compiled from 16 reports, each detailing site investigations from the late 1980s to early 1997. These reports include the 14 EPRI site investigations, plus two additional reports characterizing the comanagement of FGD sludge

with low volume wastes published by EPRI in 1994 (i.e., the "sodium-based FGD sludge" and the "calcium-based FGD sludge" reports). Appendix F presents additional waste charaterization data.

Table 3-1. Pore Water Characterization Data for Utility Coal Combustion Wastes: Facility-Averaged

Constituent	50th % Observed Concentration (mg/l)	95th % Observed Concentration (mg/l)
Antimony	(a)	a)
Arsenic	0.0973	9.64
Barium	0.136	27.4
Beryllium	(b)	(b)
Cadmium	0.00448	0.156
Chromium	0.0457	0.746
Copper	0.037	0.690
Lead	0.0138	0.468
Mercury	0.000796	0.000796
Nickel	0.0883	8.33
Selenium	0.121	1.03
Silver	(a)	(a)
Vanadium	0.157	0.800
Zinc	0.0825	23.1

Concentrations at each FFC co-management site were averaged and the resulting averages arrayed to obtain the median and high end concentrations presented in this table.
a. Concentrations for antimony and silver were not detected at any site. Therefore concentration data for these two constituents are not presented in this table.
b. As discussed elsewhere in this report, insufficient data were available for beryllium to obtain realistic median and high end concentrations.

3.1.3 Waste Management

An Electric Power Research Institute (EPRI) study of comanagement practices surveyed operators of 253 active utility coal combustion waste (CCW) management units. Of these units, 206 (81 percent) comanaged large-volume waste with at least one low-volume waste. These 206 comanagement units accounted for nearly 53 million tons (84 percent) of the 63 million tons per year of large-volume CCW reported by all active units in the survey.

The specific low-volume wastes comanaged by individual units can include any or all of the supporting process wastes listed in the previous section. They also include, in some cases, general plant waste streams, municipal wastes, asbestos, and dredged soils. Individual waste management units may comanage as many as 15 different low-volume waste streams. Typically, however, surface impoundments comanage more different waste types (a median of eight) than do landfills (a median

of four). The most frequently comanaged wastes are floor drain and sump wastes, demineralizer regenerant, coal mill rejects and pyrites, air heater or precipitator washes, coal pile runoff, and boiler blowdown.

These wastes, despite the label "low-volume," can be comanaged in quite large quantities. This is particularly true for liquid wastes, some of which can be generated at an individual facility at rates of millions of gallons per day. Accurate assessment of the total quantity of low-volume wastes is complicated because of the variation in solids content in liquid waste. Based on the EPRI data and making varying assumptions about solids content, estimates of total low-volume waste comanaged range from 5 to 53 percent of total large-volume CCW. On an individual waste management unit basis, low-volume waste comanaged can range from less than 1 percent to hundreds of times the quantity of large-volume CCW.

With the low-volume wastes, individual waste management units may comanage one or several of the four large-volume CCWs. The most common scenario is the combined management of fly ash and bottom ash with low-volume wastes in a single unit. The second most common practice is the comanagement of fly ash only with low-volume wastes.

Of the 206 comanagement units, more than half (54 percent) are surface impoundments. These surface impoundments account for only 35 percent of the large-volume CCW managed by these units. Therefore, while the number of surface impoundments and landfills are nearly equal, landfills are more significant in quantity of waste managed. In addition, the opening dates of the units in the EPRI comanagement survey reveal a trend toward the increasing use of landfills. Units opened in recent decades are more likely to be landfills than surface impoundments.

Table 3-2 presents statistics on the size of management units. According to the data, the size of each type can vary greatly. Comparing the mean and median values for each unit type suggests that units are not distributed evenly throughout this range. Figure 3-2 graphically presents the size distribution of comanaged waste landfills and impoundments. Approximately 60 percent of comanaged waste landfills and impoundments are less than 4 million cubic yards in capacity. Another

20 percent fall between 4 million and 8 million cubic yards. The remaining units are distributed over a broad range.

Table 3-2. Size of Coal Combustion Waste Management Units

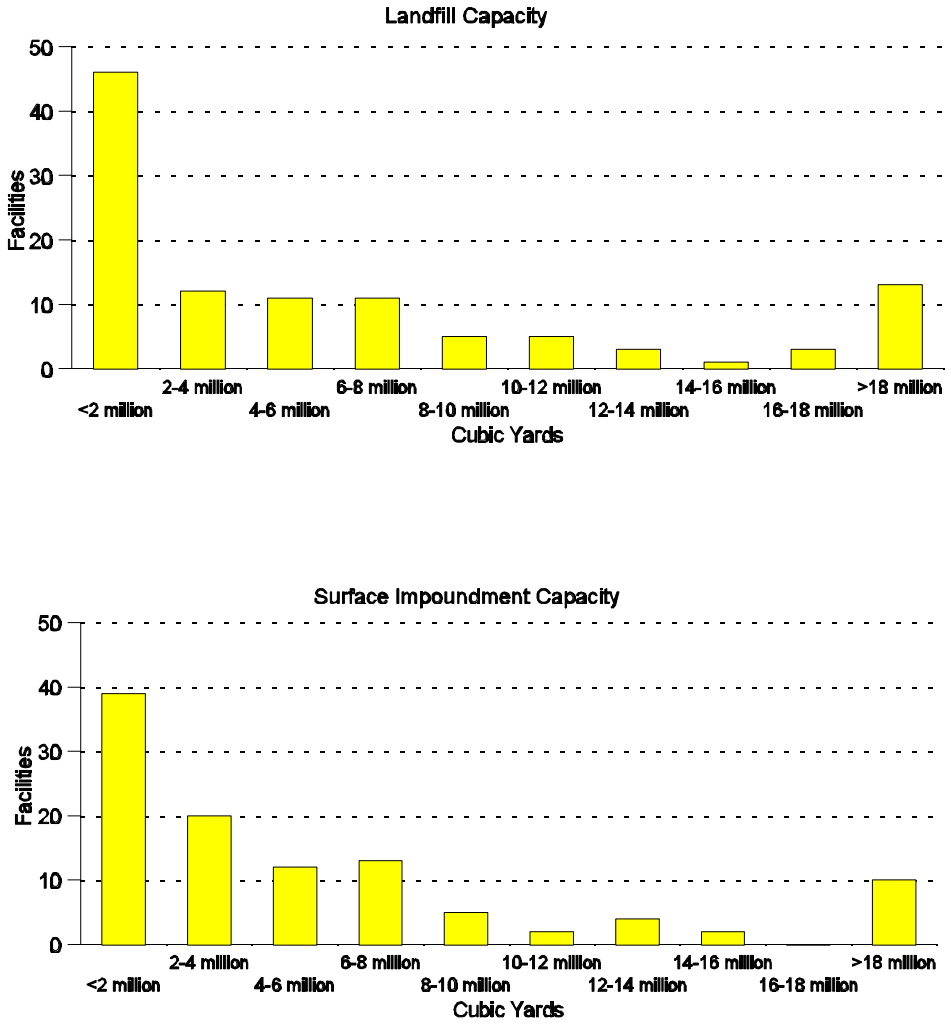
	Landfills (110 units)			Surface Impoundments (107 units)		
	Capacity (cu. yds)	Area (acres)	Height (feet)	Capacity (cu. yds)	Area (acres)	Depth (feet)
Minimum	2,700	2.6	0.36	115,000	5	1
Maximum	82,000,000	900	150[a]	63,000,000	1,500	200[b]
Median	3,850,000	66	31	3,400,000	90	20
Mean	7,434,852	116	43	6,507,405	149	36

a. One landfill yielded an estimated height of 356 feet and was omitted from this table. The data did not influence the calculated median value.
b. One surface impoundment yielded 697 feet and was omitted from this table. The data did not influence the calculated median value.
Data source: EPRI comanagement survey. Height and depth data are derived from the reported capacity and area for each unit. To increase sample size, data shown are for all active units, not just comanagement units. Dimensional data for the population of comanagement units are not significantly different from those shown.

Figure 3-3 shows the geographic distribution of CCW management units. This distribution is consistent with the distribution of coal-fired power plants shown in Figure 3-1. Figure 3-3 also shows that surface impoundments outnumber landfills in the Southeast and some Midwestern states. Landfills outnumber surface impoundments in Texas and the Southwest.

CCW comanagement units often incorporate environmental controls. Table 3-3 presents data on the types and frequency of use of environmental controls. Overall, environmental controls are employed more frequently at landfills than at surface impoundments. The types of liners reported include compacted ash, compacted clay, geosynthetic, and composite, with compacted clay being

Figure 3-2. Size Distribution of CCW Management Units



Data source: EPRI comanagement survey.

the most common. The types of covers reported are primarily soil, sand, or compacted clay, although a few landfills report geosynthetic covers. Note that the data shown for covers are for active units. For landfills, therefore, they likely reflect interim cover on completed cells or daily cover used in active cells. For surface impoundments, they probably represent covers placed on closed or filled sections of active impoundments.

Figure 3-3. Location of CCW Management Units

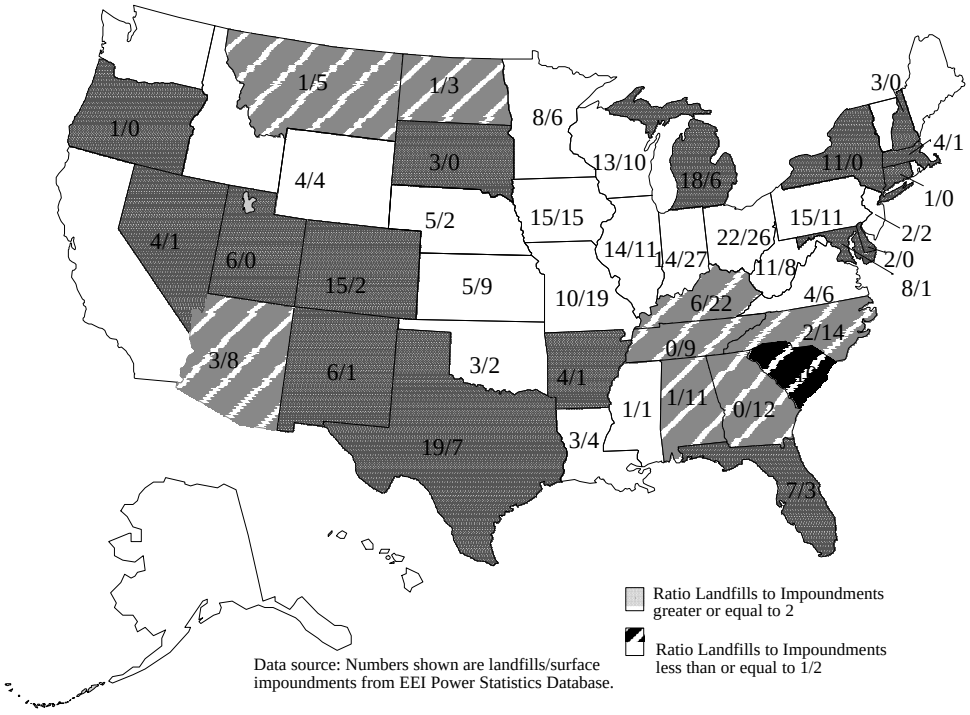


Table 3-3. Environmental Controls at CCW Comanagement Units

Environmental Control	Landfills		Surface Impoundments	
	# reporting data	% with control	# reporting data	% with control
Liner	94	57	111	26
Cover	72	94	47	30
Leachate collection	95	43	111	1
Ground-water monitoring	95	85	111	38
Ground-water performance standards	94	77	107	48
Regulatory permits	94	94	110	85

Data source: EPRI comanagement survey.

Examining cover types used on currently closed units can provide a better sense of the types of final covers that will be applied to currently active units at the end of their useful life. Fifty-three closed units provided cover information in the EPRI comanagement survey. Of these, 81 percent used some type of cover, typically soil, sand, or compacted clay.

The permitting agency for most of the units with regulatory permits is a State government. Several units are subject to the requirements of more than one permitting authority. The EPRI comanagement survey did not collect information about the substantive requirements of these permits, although it did collect information on the application of ground-water performance standards. The types of standards applied include numerical water quality standards, such as Federal maximum contaminant limits (MCLs), and nondegradation standards under which current conditions are compared with past measurements. Some units, particularly landfills, have standards tailored to the particular site. The survey did not identify the consequences if ground-water standards were exceeded. In addition, 31 (25 percent) of the units subject to ground-water performance standards are not required to monitor ground water.

In addition to traditional waste management units, large-volume CCWs are sometimes placed in minefills. The EPRI comanagement survey identified eight minefills, six of which reported comanaging large-volume CCWs and at least one low-volume waste. These minefills are located in Colorado, Missouri, Indiana, West Virginia, Michigan, Montana, Illinois, and North Dakota.

EPA also examined information on 30 minefill projects permitted in the Pottsville Mining District in central Pennsylvania. The Pennsylvania minefill projects received wastes from conventional coal combustion (e.g., pulverized coal-fired boilers), as well as fluidized bed combustion (FBC) wastes (see Section 3.3). Statistics for the Pennsylvania projects demonstrate a median minefill capacity of 917,466 cubic meters (1,200,000 cubic yards), a median surface area of 140,993 square meters (35 acres), and a median depth of 7.56 meters (25 feet).

3.2 OIL-FIRED UTILITY WASTES

3.2.1 Industry

Oil combustion by utilities is primarily a regional phenomenon, with utilities in Florida and the Northeast being the most significant consumers of oil. Utilities in other regions also use oil-fired boilers to generate power during periods of peak demand. Oil usage by utilities has shown a general declining trend, from 5.6 billion gallons in 1992 to 3.9 billion gallons in 1996 (DOE 1998b).

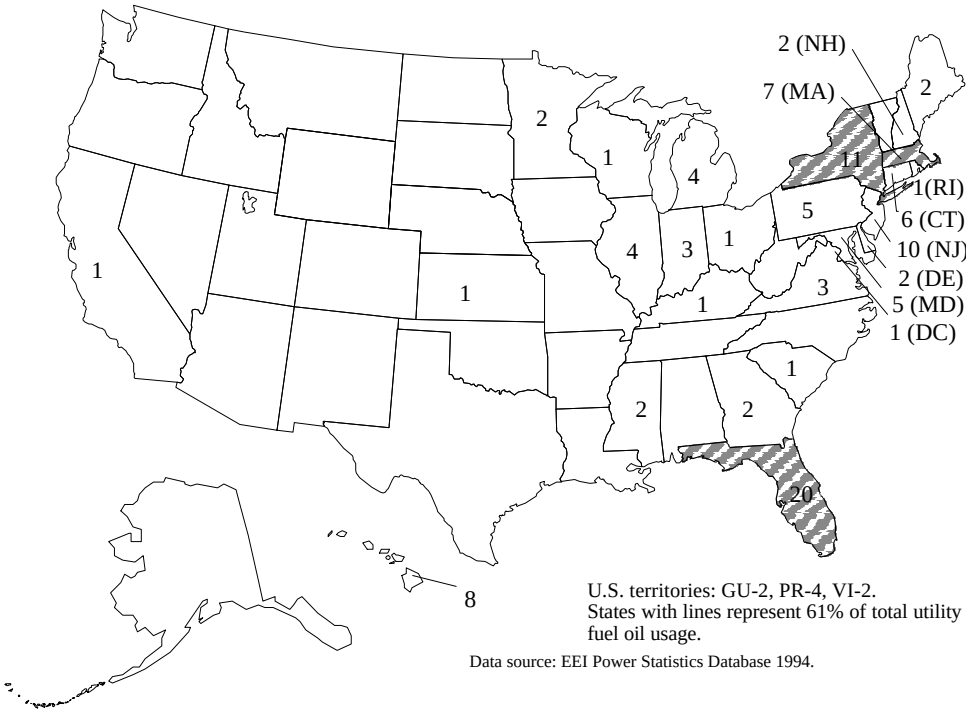
The two primary technologies for oil combustion are combustion turbines and steam-electric boilers. In combustion turbines, oil is burned inside a power-producing unit, and little or no ash is produced (EPRI 1998). Oil-fired steam-electric generators are similar in principle to coal combustion technologies. Combustion takes place in a boiler, where sufficient heat is generated and transferred to water to produce steam. The process begins with atomization of fuel oil. The atomized oil is injected into the furnace, where the burners disperse and ignite the fuel oil into a preheated air stream, thereby creating efficient fuel-air mixing (Babcock & Wilcox 1992).

Oil combustors represent a smaller portion of the utility fossil fuel combustion universe than coal combustors. One hundred fourteen oil-combusting utilities operated in the United States during 1994 (EEI 1994). Oil-fired boilers represented approximately 9 percent of the total utility fossil fuel-generating capacity for that same year. Figure 3-4 shows the distribution of oil-fired utilities throughout the United States. As noted previously, most plants are located in the Northeast and Florida. New York, Massachusetts, and Florida represent 61 percent of the total utility fuel oil usage (DOE 1998b).

3.2.2 Waste Volumes and Characteristics

The burner technology used in oil-fired steam-electric boilers can be used to combust distillate oils (e.g., No. 1, No. 2, and No. 4 fuel oils), residual oils (e.g., No. 5 and No. 6 fuel oil), and natural gas. The distillate oils and natural gas have little or no ash content, while the ash content of the residual oils is 0.009 to 0.16 percent by weight (EPRI 1998). Most of the oil burned by electric utilities is residual fuel oil. Because of the low ash content of fuel oil, many oil-fired utilities

Figure 3-4. Location of Oil-fired Power Plants



do not require particulate control equipment. Sixty-six percent of oil-fired boilers have no particulate control and, therefore, do not collect fly ash (EEI 1994). In addition, oil-fired utilities with particulate control have a control device collection efficiency lower than that of the devices used at coal-fired utilities because of the smaller particle size and lower resistivity of oil ash (EPRI 1998).

The total volume of utility oil combustion waste (OCW) is small compared with the quantity generated by the coal-fired utilities. This difference is due to the smaller number of oil-fired utilities, the low ash content of fuel oil as opposed to coal, and the lower rate of particulate collection. The total volume of utility OCW in 1995 was estimated between 15,600 and 90,000 tons, with a best estimate of 23,000 tons (EPRI 1998).

Typically, 70 percent of OCW is fly ash and 30 percent is bottom ash. With the exception of coal mill rejects, pyrites, and coal pile runoff, oil-fired utilities generate the same low-volume combustion wastes as coal-fired utilities.

Table 3-4 provides leachate characteristics for oil combustion wastes. This information comes from EPRI's oil ash database and represents a compilation of different waste types and leachate procedures. Appendix F presents additional waste characterization data.

Table 3-4. Leachate Concentrations for Oil Combustion Wastes

Constituent	50th % Observed Concentration (mg/L)	95th % Observed Concentration (mg/L)
Arsenic	0.154	4.15
Barium	0.49	12.9
Cadmium	0.085	0.62
Chromium	0.3	3.44
Copper	0.43	3.415
Lead	0.144	13.4
Mercury	0.001	0.50
Nickel	470	470
Selenium	0.0765	0.37
Silver	0.032	0.15
Vanadium	273	882
Zinc	2.35	8.12

Notes: Many constituents were not detected in one or more analyses; in such cases all measurements identified as below detection limits were assigned concentrations equal to one-half the detection limit. Constituents are presented if they were reported above detection limits in at least one waste sample.

3.2.3 Waste Management

Utility oil combustion wastes (OCWs) are managed in landfills and surface impoundments (commonly called solids settling basins in the industry). Surface impoundments, however, are rarely the final disposal unit. OCWs typically remain in surface impoundments temporarily, and then the solids are dredged and transported to an offsite landfill or used for vanadium recovery. Thus, a given OCW stream may be managed in both a surface impoundment and a landfill in the course of waste management.

EPRI surveyed 17 oil-fired utility plants (EPRI 1998). These sites report a total of 52 waste management units: 35 onsite surface impoundments, 1 onsite landfill, 14 offsite landfills, and 2 other

units.[1, 2] Table 3-5 provides statistics on the area, capacity, and depth of the surface impoundments described in the EPRI oil combustion report. The report describes total volume and capacity of all impoundments at a given site, not the area and capacity of each impoundment. Therefore, the numbers given are site totals, not individual impoundment sizes. The site totals indicate that OCW impoundments are much smaller than comparable CCW impoundments. Although the EPRI report provides no information on the size of OCW landfills, their size is expected to be smaller than comparable CCW management units. For risk assessment purposes, the size of OCW landfills was estimated using waste generation data, as described in Appendix G. Figure 3-5 shows the size distribution of OCW surface impoundments. Most units are near the mean and median values derived from the EPRI oil combustion report.

Table 3-5. Size of Oil Combustion Waste Surface Impoundments

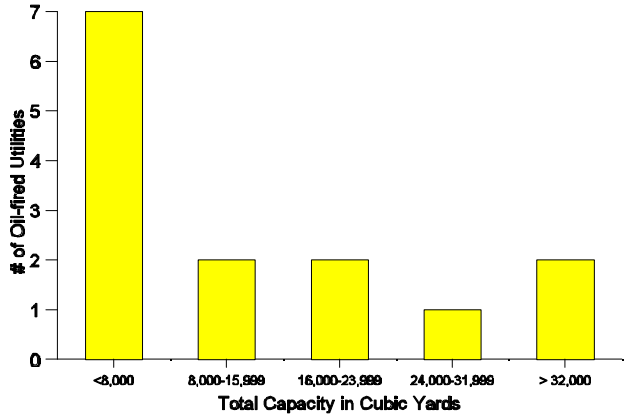
	Area (acres)	Capacity (cubic yards)	Depth (feet)
Number of impoundments	21	27	21
Mean site total	1.01	8,582	6.55
Median site total	0.90	4,951	4.74
Minimum site total	0.10	2,971	1.89
Maximum site total	2.80	29,707	21.48

Source: EPRI 1998. Depth data are derived from the reported capacity and area for each site.

[1] The two other units are a basin to which OCWs are transported dry and a pad on which OCWs stabilized with a cement-based mixture are placed.

[2] This count assumes one offsite landfill for each of the 14 sites that indicated bottom ash and/or dredged solids were taken to an offsite landfill when vanadium recovery is infeasible. The number of offsite landfills could be larger if bottom ash and dredged solids are taken to separate facilities or small if vanadium recovery is employed instead of disposal.

Figure 3-5. Size Distribution of OCW Surface Impoundments



Data source: EPRI 1998.

Of 34 surface impoundments reporting data on liner use in the EPRI oil combustion report, 18 (53 percent) have either a plastic (HDPE) or a concrete liner. Of the 16 unlined surface impoundments, 12 are located in Florida and are percolation basins designed to discharge to ground water. This practice of discharging to ground water is allowed under State wastewater permits. Of the 17 sites in the EPRI report, at least 7 monitor ground water. These sites operate 16 (46 percent) of the 35 surface impoundments, the onsite landfill, and both of the other onsite management units. This monitoring covers more than half the units where ground-water discharge is most likely (10 of the 16 unlined units and 8 of the 12 percolation basins). No data are available on liners or other environmental controls employed at the offsite landfills identified in the EPRI report. All of the offsite landfills operate under State solid waste disposal permits except one, which is a State-permitted hazardous waste landfill.

3.3 FLUIDIZED BED COMBUSTION WASTES

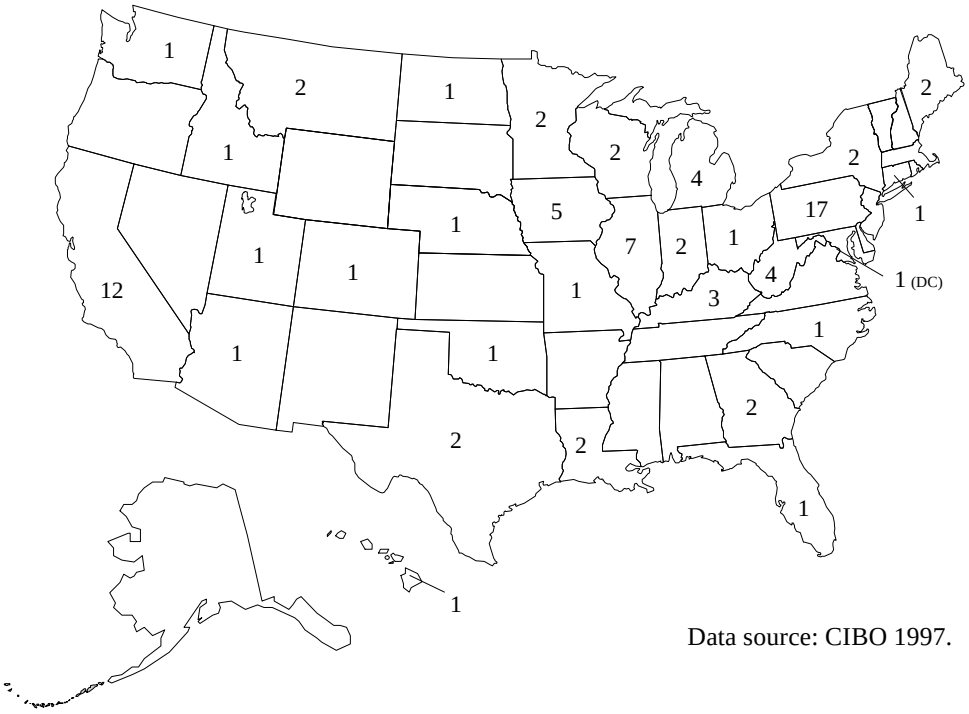
3.3.1 Industry

Fluidized bed combustion (FBC) is an emerging technology for the combustion of fossil and other fuels. While FBC combustors make up only a small part of the fossil fuel combustion universe, the increasing use of the technology, the volume of waste generated, and the differences between FBC and conventional combustion byproducts encouraged EPA to consider these wastes separately in its 1993 Regulatory Determination (58 FR 42466).

In FBC processes, fuel is burned in a bed of incombustible material while air is forced upward at high velocities, making the particles flow as a fluid. Combustion temperatures are below those used for conventional processes—bed temperatures are maintained between 1,500°F and 1,700°F. These conditions create advantages over conventional processes in terms of fuel flexibility and combustion efficiency (Babcock & Wilcox 1992). In addition, a sorbent material, typically limestone, often makes up some of the bed material. This sorbent, along with the low temperature, allows the efficient capture of sulfur oxides without the end-of-stack scrubbers required for conventional combustion processes (CIBO 1997).

As noted previously, FBC combustors make up only a small part of the fossil fuel combustion universe. Use of the technology, however, has displayed an increasing trend. In 1978, there were four plants with four FBC boilers in the United States. As of December 1996, there were 84 facilities with 123 FBC boilers representing 4,951 megawatts of equivalent electrical-generating capacity (CIBO 1997). These FBC boilers make up 1 percent of the total fossil fuel combustion-generating capacity. FBC technology is used in both the utility and non-utility sectors. The Council of Industrial Boiler Owner's (CIBO) survey of 45 FBC facilities covered five primary Standard Industrial Codes: electricity generation, food and kindred products, paper and allied products, universities, and municipal government (CIBO 1997). FBC facilities are distributed throughout the United States, as shown in Figure 3-6.

Figure 3-6. Location of FBC Facilities



Data source: CIBO 1997.

3.3.2 Waste Volumes and Characteristics

The main waste streams generated by FBC are bed ash (or bottom ash), which is spent bed material and fuel ash removed from the boiler bottom, and fly ash entrained in and removed from the exit air stream. The total FBC waste generation in 1995 is estimated between 9,091,600 and 13,150,560 tons, with a most likely estimate of 9,417,500 tons (CIBO 1997). This amount of waste equals approximately 10 percent of the waste generation from coal-fired utilities. The higher waste generation per unit reflects the inclusion of bed material/sorbent, the high ash content of some of the materials burned in FBCs (e.g., anthracite culm with an ash content sometimes greater than 50 percent), and the generally high utilization of the non-utility energy wholesalers represented in the population.

Table 3-6 summarizes leachate characteristics for FBC waste as described in the CIBO report. All concentrations are TCLP/EP. Appendix F contains additional waste characterization data.

Table 3-6. FBC Waste Characteristics - Landfill TCLP/EP Data

Constituent	50th % Observed Concentration (mg/L)	95th % Observed Concentration (mg/L)
Antimony	0.34	1.29
Arsenic	0.05	0.35
Barium	0.25	2.6
Beryllium	0.025	0.28
Cadmium	0.025	0.09
Chromium	0.039	0.29
Copper	0.07	0.16
Lead	0.05	0.49
Mercury	0.001	0.01
Nickel	0.037	0.42
Selenium	0.05	0.26
Silver	0.025	0.13
Thallium	0.05	0.07
Vanadium	0.34	1.64
Zinc	0.075	4.46
Note: Constituents reported as not detected were assigned a value by CIBO equal to one-half the detection limit.		

3.3.3 Waste Management

Waste management data are available for 23 FBC facilities that responded to either the CIBO survey or the EPRI comanagement survey. These 23 facilities reported a total of 25 waste management units: 12 onsite landfills, 5 offsite landfills, 4 onsite surface impoundments, and 4 offsite units of unknown type. These data show landfilling as the most common FBC waste management practice, accounting for 81 percent of identified FBC waste management units. On the basis of waste quantity, landfilling also is more significant, accounting for approximately 74 percent of the waste managed.[3] Because of the much greater significance of landfilling and the small sample size for impoundments, the remainder of this section focuses on describing FBC waste landfills.

[3] The four surface impoundments reported managing 550,970 tons of FBC waste in 1995. Only nine landfills provided data on the quantity managed, which totaled 828,595 tons. Assuming that the other eight landfills manage an average FBC waste disposal quantity, the quantity landfilled is an estimated 1,565,124 tons. Thus, landfilling accounts for an estimated 74 percent of the FBC waste managed.

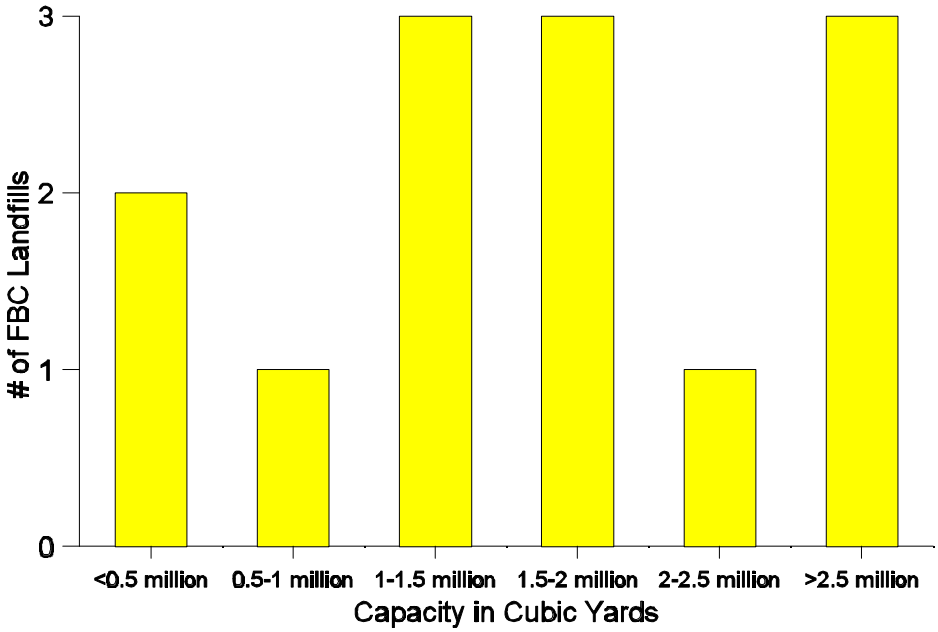
Table 3-7 presents statistics on the size of FBC waste landfills. These units are smaller than typical landfills managing conventional utility coal combustion wastes. Figure 3-7 shows the distribution of landfills by capacity for those units for which data are available. While most landfills are clustered around the median, two units in the population are much larger, with capacities of 5 million and 6.1 million cubic yards.

Table 3-7. Size of FBC Waste Landfills

	Capacity (cubic yards)	Area (acres)	Height (feet)
Number of units	13	11	10
Minimum	350,000	17	17
Maximum	6,100,000	96	75
Median	1,500,000	38	52
Mean	2,063,461	38	51

Source: responses to CIBO and EPRI comanagement surveys. Where a landfill reported two of the three dimensions, the third was estimated from the other two. If a landfill reported only capacity and area, for example, the height was calculated by dividing the reported capacity by the area.

Figure 3-7. Size Distribution of FBC Landfills



Data source: CIBO 1997.

FBC waste landfills often incorporate environmental controls. Table 3-8 presents data from the CIBO and EPRI comanagement survey responses on the types and frequency of use of environmental controls. The types of liners reported include compacted ash, compacted clay, synthetic, and composite clay and plastic. Of the landfills with regulatory permits, 27 percent have more than one permit.

Table 3-8. Environmental Controls at FBC Waste Landfills

Environmental Control	No. of Landfills Reporting Data	Percent with Environmental Control
Liner	12	42
Cover	13	62
Leachate collection	17	59
Runon/runoff control and/or collection system	14	93
Dust suppression	11	91
Compaction	11	55
Ground-water monitoring	16	77
Surface water monitoring	14	29
Air monitoring	13	8
Regulatory permits	17	89

Data source: CIBO 1997.

In addition to traditional waste management units, FBC wastes frequently are placed in minefills. Approximately 60 percent of the FBC wastes generated are used in mine reclamation (CIBO 1997). The CIBO survey responses specifically identified six minefills: four in Pennsylvania, one in Missouri, and one in Oklahoma. Section 3.1.3 presents general minefill statistics.

3.4 NON-UTILITY FOSSIL FUEL COMBUSTION WASTES

3.4.1 Industry

The non-utility fossil fuel combustion universe includes both coal- and oil-fired boilers. Steam generated by non-utility combustors is used to generate electricity, to provide heat, or as a production process input. Oil-fired boilers account for the largest fraction of non-utility fossil fuel-generating capacity (37 percent) (1990 U.S. EPA National Interim Emission Inventory, or US 90, data). Oil-

fired non-utilities make use of the same combustion technologies as utilities, but consume a higher proportion of distillate fuel oil (DOE 1998b).

Coal combustion also is significant for non-utilities, making up 22 percent of capacity (based on US 90 data). Coal-fired non-utilities also use the same technologies as utilities, although stokers make up a much larger proportion of non-utility coal-fired capacity. Stokers' small to moderate boiler sizes and fuel flexibility make them well suited to the less energy-intensive non-utility applications.

A major difference between utility and non-utility combustors is in the size of the boilers. Table 3-9 compares average boiler capacity by technology for utilities and non-utilities. With the exception of stokers, non-utility boilers are significantly smaller than comparable utility boilers. Thus, while non-utilities account for a much larger number of boilers, their total generating capacity is small compared with that of the utility sector. For example, total non-utility coal-fired capacity amounts to only 10 percent of utility coal-fired capacity.

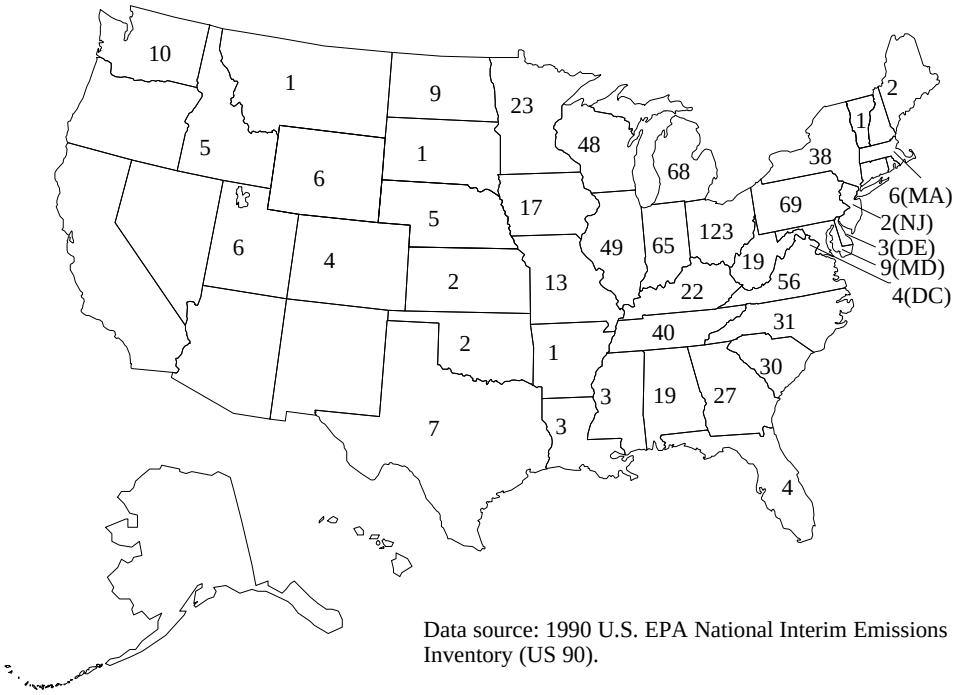
Non-utilities include an extensive and wide variety of industries and industrial activities spread throughout the United States. According to the US 90 database, industries for a significant

Table 3-9. Capacities of Conventional Non-Utility and Utility Combustors

Combustion Technology	No. of Boilers	Total Capacity (MWe)	Average Capacity (MWe)
Non-utility coal-fired	2,288	32,895	14
Non-utility Pulverizers	522	15,066	29
Non-utility Stokers	1,745	17,040	10
Non-utility Cyclones	21	789	38
Utility coal-fired	1,248	322,482	258
Utility Pulverizers	1,064	295,532	278
Utility Stokers	94	1,077	11
Utility Cyclones	90	25,874	287
Non-utility oil-fired	5,245	54,363	10
Utility oil-fired	280	40,875	146

Data sources: Utility data from 1994 EEI Power Statistics Database. Non-Utility data from 1990 U.S. EPA National Interim Emission Inventory (US 90).

Figure 3-8. Location of Non-Utility Facilities



Data source: 1990 U.S. EPA National Interim Emissions Inventory (US 90).

portion of coal-fired non-utility generating capacity include paper and allied products (SIC 26), chemicals and allied products (SIC 28), food and kindred products (SIC 20), primary metals (SIC 33), and transportation equipment (SIC 37) (SAIC 1997).

3.4.2 Waste Volumes and Characteristics

Because they use the same combustion technologies, non-utilities generate the same types of combustion wastes as do utilities. Based on data from US 90, the estimated annual waste generation rate for coal-fired non-utilities is 5,776,957 tons per year (SAIC 1997). This estimate is much smaller than the total generation of utility CCWs due to the smaller size of non-utilities and their lesser use of particulate control and FGD technologies. For oil-fired non-utilities, gaps and imprecisions in the US90 database make a reliable estimate of total waste generation using a similar methodology impossible (SAIC 1996). Although the similar total capacities of the oil-fired utility and non-utility sectors suggest similar total waste generation rates, the more extensive use of distillate fuel oil in the non-utility sector would reduce ash generation in this sector. Furthermore, differences in the use of particulate control between the two sectors could substantially affect this estimate. Because data are

not available on the prevalence of particulate control in the oil-fired non-utility sector, the significance of this impact is not known.

Generation of coal mill rejects, pyrites, and coal pile runoff is expected to be limited at coal-fired non-utilities, since the small generating capacity of the facilities does not promote onsite processing of coal or require storage of large volumes of coal. In addition, the generation of coal mill rejects and pyrites as a result of coal pulverization does not occur in stokers (stokers do not have pulverizers), thereby reducing the amount of these low-volume wastes. The other types of low-volume wastes discussed in the previous sections, however, are generated by non-utilities, although in smaller quantities consistent with their smaller unit sizes. In some cases, non-utilities may generate insignificant quantities of some of these wastes. For example, boiler fireside cleaning may take place so infrequently at a small non-utility combustor that the amount of cleaning waste generated over the life of the facility may be insignificant. In addition to low-volume combustion wastes, non-utility combustors generate a wide range of non-combustion process wastes, consistent with the variety of industries represented. These process wastes also may be codisposed with combustion wastes, as described in the following section.

Detailed sampling data are not available on the characteristics of non-utility combustion wastes. For purposes of this report, non-utility combustion wastes were modeled using data for comanaged coal combustion wastes (see Section 4).

3.4.3 Waste Management

In addition to surveying FBC facilities, the Council of Industrial Boiler Owners (CIBO) has conducted a limited survey of conventional non-utility combustors. Some of the respondents to the CIBO non-utility survey provided waste management data. In addition, EPA has collected regulatory permit file information on the management of fossil fuel combustion wastes from selected non-utilities in six states (Illinois, North Carolina, Virginia, New York, Pennsylvania, and Wisconsin).

Twenty-seven FFC waste management units are covered by the CIBO non-utility survey, including 25 landfills and 2 surface impoundments. The State permit file data also support the observation that landfilling is the primary practice for non-utility combustion wastes. Of the 49 units

identified in five States, 43 (88 percent) were landfills. Because of the much greater significance of landfilling and the small sample size and limited data for impoundments, the remainder of this section focuses on describing non-utility waste landfills.

Non-utility combustion waste landfills include both onsite and offsite units. Because the review of State permit fill information was directed at facilities operating captive disposal facilities, the landfills identified by this effort are all onsite units. However, other information gathered during the review of permit files point to the conclusion that the majority of non-utility combustion wastes are managed offsite in at least three of the six States studied (Illinois, New York, and Pennsylvania) (SAIC 1997). Neither the State permit files nor the CIBO non-utility survey provided information on the size of non-utility combustion waste landfills. Given the lower waste generation rates for non-utilities, however, these units are expected to be much smaller than utility landfills. For risk assessment purposes, the size of non-utility landfills was estimated using waste generation data, as described in Appendix G.

Based on the CIBO non-utility survey, comanagement appears to be as common at non-utilities as at utilities. Thus, the selection of comanaged waste characterizations on data for non-utilities is appropriate. Of 22 units for which data are available, 19 (86 percent) comanage combustion wastes with low-volume wastes. Nine of these 19 comanage with low-volume combustion wastes, 3 comanage with other process wastes, and 7 comanage with both. In general, the types of low-volume combustion wastes managed at non-utilities are similar to those managed at utilities. The CIBO non-utility survey did not collect information on the specific types of other process wastes comanaged with non-utility combustion wastes. The State permit file information also generally reveals that comanagement is common at non-utilities. In two of the three States for which comanagement information is available, 90 percent or more of the units comanage combustion and other wastes (9 of 10 landfills in North Carolina and 17 of 18 landfills in Wisconsin). On the other hand, in the third state, Pennsylvania, most non-utility combustion waste is managed separately.

Table 3-10 presents data on the types and frequency of use of environmental controls at non-utility combustion waste landfills. The CIBO non-utility survey and the State permit file information lead to differing conclusions about the percentage of landfills that are lined. It is unclear which

sample is more representative, although the State permit file units are not geographically representative and are probably biased toward having liners, as they were identified by examining permit-related information. Similarly, the State permit file data show the use of leachate collection, runoff controls, and ground-water monitoring to be more common at non-utilities than at utilities. It is probable, however, that the State permit file sample is biased toward the use of all three of these environmental controls.

Table 3-10. Environmental Controls at Non-Utility Combustion Waste Landfills

Environmental Control	No. of Landfills With Data	Percent With Control	Data Source
Liner	19	16	CIBO non-utility survey
	27	52	State permit files
Leachate collection	30	67	State permit files
Runoff control	6	100	State permit files
Ground-water monitoring	34	94	State permit files
Regulatory permit	29	52	CIBO non-utility survey

4. DEFINITION OF WASTE MANAGEMENT SCENARIOS

4.1 GENERAL CONSIDERATIONS

As described in Section 2, the U.S. Environmental Protection Agency (EPA) used EPACMTP to estimate potential risks to human health from the ground-water exposure pathway from the management of fossil fuel combustion (FFC) remaining wastes. The analysis of these risks included three basic steps: (1) identifying scenarios of most concern, (2) quantifying model input parameters based on these scenarios, and (3) conducting model runs and evaluating the results. Section 4.2 identifies waste management scenarios of concern based on the general analysis of management methods presented in Section 3. It then discusses the methodology and logic used in developing input parameters for these scenarios. Later sections of this report present and evaluate modeling results.

4.1.1 Calculating Benchmarks

Table 4-1 presents principal assumptions regarding the receptors. Note that central tendency assumptions were used throughout. This model did not consider showering as an exposure because showering results in exposure from inhalation, which is not applicable for metals in this context.

Table 4-1. Assumptions for Adult Residents (Both Sexes) Scenario, Ground-Water Ingestion Pathway

Exposure Parameter	Value Used	Source
Ingestion rate (L/d)	1.4	Mean ingestion rate[a]
Exposure duration (yr)	9	Median residence time[b]
Body weight (kg)	72	Mean body weight[c]
Lifetime (yr)	75	Mean life expectancy[d]
Exposure frequency (days/yr)	350	Assumed

Source: *EPA Draft Exposure Factors Handbook*, 1996.
a. Derived from Table 3-10.
b. Derived from page 14-16.
c. From Table 7-10.
d. From page 8-1.

EPA combined the adult receptor exposure assumptions above with toxicological information for the constituents of concern into benchmark values. Table 4-2 shows the results (See Appendix B for a more detailed derivation of these values. Note that in the absence of toxicity data in EPA's Integrated Risk Information System (IRIS) database or HEAST database, primary MCLs developed

under the Safe Drinking Water Act were used for the benchmark.). A different set of benchmarks was developed for assessment of risks to children. These benchmarks are presented in Section 6.

Table 4-2. Benchmark Values Derived for the Remaining FFC Waste Risk Assessment

Constituent	RfD (mg/kg/d)	Carcinogen Slope Factor $(mg/kg/day)^{-1}$	Health-Based Number (mg/L)		MCL or Action Level (mg/L)
			RfD-Based	CSF-Based	
Antimony	0.0004	–	***0.021***	–	0.006 (1 MCL)
Arsenic	0.0003	1.5	0.015	***0.00029***	0.05 (1 MCL)
Barium	0.07	–	***3.60***	–	2 (1 MCL)
Beryllium	0.005	4.3	0.26	***0.0001***	0.004 (1 MCL)
Cadmium	0.0005	–	***0.026***	–	0.005 (1 MCL)
Chromium VI	0.005	–	***0.26***	–	0.1 (1 MCL)
Copper	–	–	–	–	1.3 (action level) 1.0 (2 MCL)
Lead	–	–	–	–	0.015 (action level)
Mercury	0.0003	–	***0.015***	–	0.002 (1 MCL)
Nickel	0.02	–	***1.03***	–	–[4]
Selenium	0.005	–	***0.257***	–	0.05 (1 MCL)
Silver	0.005	–	***0.257***	–	0.1 (2 MCL)
Thallium	0.00008	–	***0.0041***	–	0.002 (1 MCL)
Vanadium	0.007	–	***0.360***	–	–
Zinc	0.3	–	***15.4***	–	5 (2 MCL)

For carcinogens, the health-based number (HBN) is calculated from the following equation:

$$HBN = \{risk \times BW \times AT \times 365\}/\{I \times ED \times EF\},$$

where $risk = 10^{-6}$

For non-carcinogens (i.e., all constituents with RfDs), the health-based number is calculated from the following equation:

$$HBN = \{HQ \times BW \times RfD\}/I,$$

where $HQ = 1$.

EPA generated a list of waste management scenarios for ground-water modeling for each of the four FFC waste generating sectors. Table 4-3 presents this list. Each of the scenarios has been assigned a two-letter identifier for ease of reference throughout this report.

4.1.2 Identifying Scenarios

Table 4-3. Scenarios Modeled in EPACMTP for Ground-Water Risk Assessment

FFC Waste Type	Management Scenarios
Coal-fired utility comanaged wastes	Surface impoundment . . . . . Scenario CS Onsite landfill . . . . . Scenario CL Minefill . . . . . Scenario CF
Oil-fired utility wastes	Surface impoundment . . . . . Scenario OS Onsite monofill . . . . . Scenario OM Commercial landfill . . . . . Scenario OL
FBC wastes	Onsite landfill . . . . . Scenario FL Minefill . . . . . Scenario FF
Non-utility combustion wastes	Onsite monofill . . . . . Scenario NM Commercial landfill . . . . . Scenario NL

Two major factors influenced EPA's identification of waste management scenarios for modeling. First, EPA considered only those scenarios of principal concern from a ground-water standpoint. For example, the practice of beneficially using FFC wastes as soil amendments is well known. This practice was assessed in the above-ground multi-pathway human health and ecological risk assessment. Although this management practice could also present risk via ground-water, it is likely to result in ground-water risks that are lower than those from a landfill scenario. Therefore, the use of FFC waste as soil amendment was not included in ground-water modeling.

The second factor EPA considered was whether the practice actually occurs. For example, oil-fired utility wastes are known to be managed in commercial offsite landfills, as documented in EPRI's oil combustion report (EPRI 1998). Therefore, such a scenario was considered appropriate for assessing risks from oil combustion wastes. However, coal-fired utility comanaged wastes are managed in onsite (or captive offsite) units operated by the utility and are not mixed with wastes generated from other facilities. Therefore, a commercial landfill scenario was not considered appropriate for comanaged coal combustion wastes.

Each scenario identified in Table 4-3 represents one of four waste management unit types (surface impoundment, onsite landfill, commercial landfill, or minefill). General considerations associated with each waste management unit type are discussed below. The following considerations are applicable for all wastes evaluated for the given unit type:

- ***Onsite surface impoundment***—Onsite surface impoundments were assumed to contain only remaining FFC wastes. EPACMTP modeled the fate and transport of leachate released during the active lifetime of the waste management unit. During its operational life, the impoundment was assumed to release leachate at a constant rate and at a constant concentration of contaminants. At the end of the impoundment's operational life, it was assumed leaching would cease and all wastes would be removed from the impoundment. The model continued to track containment plume migration for the entire study period (10,000 years).

- ***Onsite landfill***—Onsite landfills were assumed to contain only FFC wastes. EPACMTP landfill leaching behavior was time dependent. During the operational life of the unit, it was assumed that no leaching would take place. At the end of the active life, it was assumed that the landfill would be closed and capped with native/local materials. Leaching would then begin at the end of the active life of the unit at a constant infiltration rate and an initial leachate concentration. Over time, the leachate concentration would decrease (although the infiltration rate would be constant), until all of the contaminants leached out of the unit or the end of the study period (10,000 years) was reached.

- ***Commercial landfill***—These landfills were assumed to manage FFC wastes in conjunction with other, unrelated wastes. EPACMTP model runs considered only the incremental risk associated with FFC wastes and did not consider any aspect of the other wastes (e.g., synergistic effects). Other aspects of the commercial landfill were identical to the onsite landfill.

- ***Minefill***—EPACMTP models the minefill scenarios as if they were identical to the landfill scenarios. In general, this scenario best simulated a surface mine backfill project. This scenario would not be applicable to the filling of mine shafts and similar channelized flow conditions or to the co-placement or mixing with organic materials prior to mine site revegetation/reclamation. Minefill-specific waste management unit dimensions and geographic distribution were developed to distinguish these projects from the sector-specific landfill scenarios.

4.1.3 Identifying Input Parameters

All model input parameters were based on a combination of industry and waste specific data and generic data developed as part of the 1995 Hazardous Waste Identification Rule (HWIR)

proposal (60FR 66343, December 21, 1995). Some of the more significant input parameters are the following:

- ***Concentration data***—Three sector-specific databases were developed to characterize leachate from the wastes: one for coal-fired utility comanaged wastes, one for oil-fired utility wastes, and one for FBC wastes. The coal-fired utility data were used in assessing the non-utility combustion waste scenarios due to a lack of non-utility specific characterization information. Each of the three databases was based on information collection efforts conducted by industry following the 1993 regulatory determination.

  Concentration data used as input parameters for modeling were calculated as follows for coal-fired utility comanaged wastes and oil-fired utility wastes. First, all waste characterization data (i.e., all samples) for a given site were averaged to arrive at a single leachate concentration. A concentration of one-half of the reported detection limit was used for samples whose concentrations were at or below the detection limit. Then, the average values for sites in each scenario were rank-ordered, and EPA selected as the model input parameter the concentration below which fell at least 95 percent of all values--when there were fewer than 20 values (that is, when there were data for fewer than 20 sites), EPA selected the highest value. For fluidized bed combustion wastes, EPA used data provided by the Council of Industrial Boiler Owners, as described in Section 4.4 below.

- ***Management unit characterization data***—Input parameters specific to the waste management units modeled were based in whole or in part on data specific to the FFC waste industry sector under consideration. Where appropriate, some unit-specific parameters were based on more universal (rather than specific FFC waste) waste management industry data. For example, the non-utility commercial landfill (Scenario NL) incorporated a typical landfill size estimated from the 1986 Industrial D Waste Management Survey, as reported in the EPACMTP Users Guide (EPA, 1995c). The specific data used to develop unit-specific model inputs for each scenario are described further in the sections below.

- ***Management unit locations and ground-water classifications***—EPA used available data on the geographic location of FFC waste management unit locations to develop input values for infiltration rate, recharge rate, and correlated ground-water input parameters (e.g., hydraulic gradient, hydraulic conductivity). Specifically, values for infiltration and recharge rates developed for HWIR using the HELP model were assigned to FFC waste management sites based on site location information. In addition, for purposes of Monte Carlo modeling, all FFC waste management unit locations were assigned to 1 of 12 ground-water classifications developed for HWIR. Both the HELP model calculations and the assignment of groundwater classifications used available information about the geographic distribution of FFC waste management units. Because data on geographic location were limited to the State level for some populations, developing these ground-water input values required simplifying assumptions. These assumptions are described in Appendix E.

- ***Receptor well location***—For the deterministic high-end analysis, EPA assumed the receptor well was located at the plume centerline, 150 meters downgradient of the management unit.

In many cases there were multiple available sources for a given data element. For example, comanagement unit location data are available from both the EPRI site investigations and the EPRI comanagement survey. In every case, the source chosen reflects consideration of the appropriateness of the data, its representativeness of the industry, and its accuracy. The following sections further define each of the modeled waste management scenarios and describe the selection of corresponding model input values.

Table 4-4 shows the input parameters used in ground water modeling. (Note: Table 4-4 is located at the end of this chapter.) The specific sources for each data element and other information on the data are provided in Appendix A.. The Monte Carlo runs used a full array of the available data, while the deterministic runs generally used median values from the same arrays. Two of the parameters, however, were set to high end in all deterministic analyses. As noted above, the concentration data used in the deterministic runs corresponded to the 95th percentile concentration, while the well location parameters used in the deterministic runs corresponded to a location at the high end of the employed Monte Carlo distribution (150 meters downgradient of the management unit).

4.2 COAL-FIRED UTILITY COMANAGED WASTES

Section 3 shows landfills and surface impoundments to be the predominant onsite management practices associated with comanaged wastes. Specifically, slightly more than half of the management units identified by the EPRI comanagement survey (described in Chapter 3) are impoundments, and the remainder are landfills. Another management practice of concern is "minefilling," where wastes are placed in mine sites.

These three scenarios were selected for modeling. In all cases, simplifying assumptions were used to facilitate the modeling. Many of the major assumptions used to develop model input

parameters are briefly discussed in this section. All input parameters and data sources are provided in Appendix A.

4.2.1 Surface Impoundment (Scenario CS)

This scenario represents comanagement of coal-fired utility combustion waste (i.e., ash) with low-volume wastes in an onsite surface impoundment, a practice observed at a large percentage of coal-fired utilities, as discussed in Section 3. Based on review of the site-specific EPRI site investigation reports, the results of the comanagement survey, and observations from site visits, there are important operational characteristics that influence the assumptions made in modeling. Several of the more significant assumptions and variables are briefly discussed here.

- The modeled impoundment was assumed to receive ash and low-volume wastes for a lifetime of 40 years, at which time the wastes were removed. Leachate released from the impoundment during its life was then modeled for a period up to 10,000 years. Thus, the model would not account for a post-closure period that may have been preceded by impoundment drainage and stabilization/reclamation.

- The impoundment was assumed to be rectilinear in shape, and ash depth was based on the midpoint of the unit's life (that is, depth after 20 years). Assumed unit size, ponding depth, and liner thickness were based on the EPRI comanagement database (see Appendix G). The deterministic model used median values for impoundment size while the probabilistic approach used the full distribution of values.

- For the deterministic model, the impoundment was assumed to have a uniform depth of ash with no other liner, and this ash was assumed to have an unvarying vertical hydraulic conductivity. A similar assumption was that of constant thickness of comanaged waste that was uniformly mixed, topped by a constant depth of water during the active life, and thus a constant hydraulic head (based on ponding depth). Thus, the model would not account for either spatial or temporal variability in depth of waste or water, the type/mixture or characteristics of wastes, or other characteristics of the impoundment. The Monte Carlo approach varied some but not all of these parameters.

- The same concentration data were used for all comanagement scenarios. These data were provided by EPRI and represent samples of interstitial pore water taken from impoundment and landfill waste core samples. A smaller number of EPRI TCLP and EP data points were not considered.

- The distribution of impoundments across the country was taken from the EPRI comanagement survey and the EEI database, as described in Section 3. Most parameter values describing the environmental conditions at these sites were taken from data developed for HWIR (most are based on data points common to the State in which the

impoundments are located), with some values taken from utility industry-specific data (EPRI 1984). Appendix E describes the methodology for calculating infiltration rate, saturated zone thickness, and many other parameters.

4.2.2 Landfill (Scenario CL)

Based on Section 3, comanagement of coal-fired utility combustion wastes and low-volume wastes in on-site landfills is equally common as comanagement in surface impoundments. Some of the observations described above for impoundments also apply to landfills, but others are unique to landfills. Several key model data elements and EPA's assumptions are discussed below.

- As noted above, concentrations were based on interstitial pore water from waste core samples. Most waste core samples were taken from impoundments, with only a few from landfills (there were too few to be representative). In general, impoundment samples showed higher concentrations than landfill samples.

- The landfill was assumed to be square in shape. It was assumed that the landfill would not generate any leachate during its 40-year life, but would generate leachate based on a constant infiltration rate for up to 10,000 years following closure. Further assumptions included a constant height or depth of waste, and uniform waste concentrations.

- Infiltration and recharge rates were also assumed to be equal. Principal assumptions included the absence of a liner, a cap of constant thickness and composition, and an assumed hydraulic conductivity for the waste in the landfill. The assumption regarding the liner is true for nearly half of all comanagement landfills (as reported in Section 3). The active period of the unit represents time when the infiltration rate is changing due to the development of a final cap, so the actual infiltration rate during this period of time is probably greater than following cap construction. However, the infiltration rate is assumed to be constant and does not account for changes in waste characteristics or other variables. The model did not allow waste hydraulic conductivity data to be used--in some cases, it is known to be quite low.

- Landfill area and capacity assumptions were based on the EPRI comanagement database (See Appendix G).

- The distribution of landfills across the country was taken from the EPRI comanagement survey and the EEI database, as described in section 3. Most parameter values describing the environmental conditions at these sites were taken from data developed for HWIR (most are based on data points common to the State in which the landfills are located), with some values taken from utility industry-specific data (EPRI 1984). Appendix E describes the methodology for calculating infiltration rate, saturated zone thickness, and many other parameters.

- The source of contaminants was assumed to be finite. That is, leaching of constituents continued for either 10,000 years or until the entire supply was exhausted, whichever came first.

4.2.3 Minefill (Scenario CF)

As noted in Section 3, coal-fired utility comanaged wastes are sometimes used/disposed at mines. Therefore, this scenario was developed to represent that practice. As noted above, the scenario most closely resembles simple placement of the wastes in a surface mine; it does not resemble backfilling underground mines or use in site reclamation. The scenario uses many of the same assumptions as the coal-fired utility comanaged waste landfill, but with several key input values modified where needed. Several of these assumptions are discussed below.

- Unlike the other impoundment and landfill scenarios, input parameters for meteorological and hydrogeologic conditions in this scenario do not reflect the geographic distribution of coal-fired utilities. Instead, EPA modeled minefilling in eight States with significant coal mining activity: Wyoming, Illinois, Indiana, Kentucky, Ohio, West Virginia, Pennsylvania, and Maryland. As is the case with the other scenarios, input parameters for meteorological and hydrogeologic conditions were taken from data developed for HWIR (based on data points common to these States).

- As noted above, concentrations were based on interstitial pore water from waste core samples. None of the comanaged waste samples were taken at minefills.

- EPA based the estimate of unit design (area, depth) on general project-specific information from 27 minefill projects in Pennsylvania. The deterministic model used median values from the Pennsylvania minefill survey. As with the landfill scenario, EPA did not explicitly consider compaction of waste and the potential influence on hydraulic conductivity and hence infiltration rate. The procedure to calculate area is presented in Appendix G.

4.3 OIL-FIRED UTILITY WASTES

EPA selected three oil-fired utility scenarios: a surface impoundment, an onsite monofill, and a commercial landfill. In all cases, simplifying assumptions were used to facilitate the modeling. Many of the major assumptions used to develop model input parameters are briefly discussed in this section. All input parameters and data sources are provided in Appendix A.

4.3.1 Surface Impoundment

As discussed in Section 3, because of the common use of solids settling basins at oil-fired utilities, a given oil combustion waste stream may be managed in both a surface impoundment and a landfill in the course of waste management. This scenario represents the first step in this management process -- the temporary storage of oil combustion wastes in an on-site solids settling basin. Many of the assumptions used for modeling the comanaged waste impoundment were also used here.

- The modeled impoundment was assumed to receive ash and low-volume wastes for a lifetime of 40 years, with periodic removal of wastes. This is a more realistic assumption for the present scenario than it is for the comanagement scenario--settling basins for oil-fired wastes are generally dredged periodically through their lives. Leachate released from the impoundment during its life was then modeled for a period up to 10,000 years.

- The impoundment was assumed to be rectilinear in shape, and ash depth was based on the reported dredging frequency and annual generation rate. Assumed unit size, ponding depth, and liner thickness were based on EPRI data (see Appendix G). The deterministic model used median values for impoundment size while the probabilistic approach used the full distribution of values.

- For the deterministic model, the impoundment was assumed to have a uniform depth of ash with no other liner, and this ash was assumed to have an unvarying vertical hydraulic conductivity. A similar assumption was that there was constant thickness of waste, topped by a constant depth of water during the active life, and thus a constant hydraulic head (based on ponding depth). Thus, the model would not account for either spatial or temporal variability in depth of waste or water, the type/mixture or characteristics of wastes, and other characteristics of the impoundment. The Monte Carlo approach varied some but not all of these parameters.

- The same concentration data were used for all oil-fired scenarios. It would not be expected that waste concentrations would vary based on the type of management, so this should not introduce any extra uncertainty into the results. The data were provided by EPRI and represent extracted leachate samples from both “as generated” wastes and surface impoundment wastes.

- Oil combustion utilities are predominantly located along the east coast, principally in the northeast and southeast. EPA used various input parameters that vary by region, including recharge rate and factors affecting groundwater velocity. These parameters could be expected to be different for these two areas of the country than for the U.S. as a whole, so each site was assigned a unique set of characteristics based on its location. The characteristics assigned were the same as used in HWIR. For the deterministic analysis, these characteristics were weighted to obtain an industry-wide median value for

each affected parameter (such as recharge rate). In the Monte Carlo Analysis, each site was assigned a meteorological and hydrogeologic condition consistent with those assigned for its geographic location in HWIR. Appendix E describes the methodology for calculating infiltration rate, saturated zone thickness, and many other parameters.

- For the remaining EPACMTP input parameters, EPA used data specific to the observed population of oil-fired utility surface impoundments whenever possible. For example, EPA assigned a value for recharge rate that accounted for the geographic distribution of surface impoundments in the EPRI oil combustion report. Values for other aquifer characteristics were based on EPRI site investigations and calculated by examining only east coast facilities, consistent with the concentration of oil combustion in that region.

- The hydraulic conductivity of the waste "liner"was assumed to be the same as used for the coal comanagement impoundment. The liner represents the layer of settled solids at the base of the impoundment.

4.3.2 On-site Monofill (Scenario OM)

As discussed in the oil impoundment scenario, solids may be dredged from impoundments and then placed in landfills. Alternatively, certain oil combustion wastes may be sent directly to a landfill. As discussed in Section 3, one of the oil-fired utilities in EPRI's oil combustion report operates an on-site landfill dedicated solely to oil combustion wastes (i.e., a monofill). Another plant operates a dry on-site ash basin, and a third facility reports use of a cement-stabilized ash pad. Similar monofills are expected to exist at other plants not covered by the EPRI report.

- EPA used the median waste generation rate derived from data reported in the EPRI report. This generation rate would include all waste generated at the facility, not just material actually reported as landfilled. A single oil combustion waste quantity was determined for each facility, and each facility was assumed to dispose of this material at its own landfill. The dimensions of this landfill were derived from this generation rate using a 30 year operating life. Details of these calculations are presented in Appendix G.

- EPA assumed that the monofill was not lined. The geographic distribution was assumed to be the same as for oil-fired waste impoundments.

- Most other input parameters were either the same as for other landfills or were derived in the same manner (see Appendix E).

4.3.3 Commercial Landfill (Scenario OL)

This scenario represents management in a commercial landfill. As discussed in Section 3, wastes dredged from oil-fired utility solids settling basins are often transported to a commercial landfill. In addition, bottom ash from oil-fired boilers may be taken directly to a commercial landfill.

- EPA assumed the landfill area was the same as for the non-utility commercial landfill scenario (section 4.5.2). Thus, the size specified was a median value for landfill sizes (area and depth) from a segment of the HWIR database.

- EPA assumed a waste fraction of 24 percent, based on the median waste generation rates used for the oil monofill scenario and the commercial landfill size derived as noted above. Each generator was assumed to use a different landfill.

- EPA assumed the landfill was unlined.

- Landfills were assumed to be near the generating sites, and then most input parameters for environmental conditions were based on HWIR values for the estimated distribution of landfills. These were based on climatic conditions at utilities in the EPRI oil combustion report, derived in the same manner as for other landfills (see Appendix E).

- The model assumes that leachate data for the FFC wastes describes the leachate from the entire unit.

4.4 FBC WASTES

EPA selected two FBC waste scenarios: a landfill and a minefill. In all cases, simplifying assumptions were used to facilitate the modeling. Many of the major assumptions used to develop model input parameters are briefly discussed in this section. All input parameters and data sources are provided in Appendix A.

4.4.1 Landfill (Scenario FL)

As discussed in Section 3, landfilling is the primary management practice employed for FBC wastes. This scenario represents an on-site FBC waste landfill and is similar in most respects to the coal-fired utility comanagement landfill scenario. Other assumptions specific to FBC wastes are noted below.

- The same concentration data were used for all FBC waste management scenarios. It would not be expected that waste concentrations would vary based on the type of management, so this should not introduce any extra uncertainty into the results. The data were provided by the Council of Industrial Boiler Owners (CIBO) from their survey of FBC facilities (CIBO 1997) and represent extracted leachate samples.

- Concentration data were not averaged by site. Rather, EPA used data provided in the CIBO report (CIBO 1997). That report presents data summary tables of EP and TCLP results for each of three wastes. A single waste type (combined ash) was assumed because most FBC waste is combined when managed; however, data for all three waste types (fly ash, bottom ash, and combined ash) were considered in developing characteristics of "combined ash" for this scenario. Unlike the coal co-management and oil combustion data, the results were not averaged by facility. Instead, an implicit assumption was made that the sampling data reflect the overall population of landfilled wastes, and leachate composition is not variable. This assumption was not tested and would be affected by variability in feed or operating conditions over time. A complete discussion of the methodology used in considering concentration data is presented in Appendix F. As with other data, non-detects were assumed to have a value equal to one-half their detection limit.

- Median values (in the deterministic analysis) for landfill area and depth were derived from values reported by respondents to the CIBO survey. Thus, this scenario specifically representative of the observed population of FBC waste landfills.

- As in other landfill scenarios, a finite source assumption and a leaching period up to 10,000 years is used. The total quantity of contaminant that can migrate from the unit is limited to the total quantity available.

- The landfill is assumed to be unlined, as is reported in Section 3 for over half of FBC landfills. The methodology for calculating input parameter values for infiltration rate, saturated zone thickness, and other parameters is given in Appendix E. In general, this methodology was similar to that used developing the parameters for coal comanagement landfill sites, except the actual locations of the FBC landfills were used instead of the locations of coal comanagement landfill sites. This methodology used data specific to the observed population of FBC waste landfills for the remaining EPACMTP input parameters whenever possible. For example, the value for recharge and infiltration rates accounted for the geographic distribution of landfills in the CIBO survey.

4.4.2 Minefill (Scenario FF)

As noted in Section 3, minefilling is even more common for FBC wastes than for coal combustion wastes. This scenario is exactly the same as that for the coal combustion waste minefill, except that it uses FBC waste concentrations (see above for assumptions regarding FBC waste concentrations).

4.5 NON-UTILITY COMBUSTION WASTES

The non-utility landfills identified in examining state permit files were all located at the generator sites, as reported in Section 3. These on-site non-utility landfills are likely to comanage combustion wastes with other FFC wastes or other (non-FFC) process wastes. As an alternative management practice, information gathered during the review of state permit files also suggested that the majority of non-utility combustion wastes in at least three States are managed in off-site landfills. Such off-site management is a likely practice in other States as well, particularly for small facilities.

Two scenarios were used to evaluate non-utility wastes: an onsite monofill scenario and a commercial landfill scenario to represent codisposal of FFC and non-FFC wastes. In general, there is little industry-specific data for non-utility wastes. Except as noted below, data and assumptions used for the coal-fired utility waste comanagement scenarios are used here as well.

4.5.1 On-site Monofill (Scenario NM)

- The deterministic analysis used a median waste generation rate derived from data reported in the US90 database. This waste was assumed to be disposed in a single onsite landfill (with no other wastes) for a period of 30 years. The dimensions of this landfill (area and depth) were derived from this accumulated volume by assuming a square, flat-topped pyramid. A more detailed description of the methodology is presented in Appendix G.

- Waste quantities were estimated from the US90 database. These in turn reflect data from the mid-1980's to early 1990's. More significantly, this source represents the larger facilities burning fossil fuels and generally excludes small generators. Therefore, use of the US90 database for this scenario may have caused a skewing of the actual distributions of generated waste quantities, resulting in the modeled waste quantities overestimating the actual quantities.

- Areas calculated as less than 3,000 square meters were eliminated from the Monte Carlo distribution because the occasional excessively small (and unrealistic) areas resulted in errors during running of the model; therefore the distribution used in the Monte Carlo analyses was slightly more conservative than the distribution used to derive the median for the deterministic analysis.

- EPA assumed the landfill was unlined, consistent with the observation (reported in Section 3) that a large percentage of non-utility landfills are unlined. As with other scenarios, EPA did not make explicit assumptions regarding factors affecting infiltration rate, but instead used the infiltration rates developed for HWIR. Among other

assumptions, those calculations assumed a two-foot soil cover (with soil composition the same as the surrounding area) and a grass vegetation cover. The recharge rate of the surrounding area was assumed to be equal to the calculated infiltration rate, as recommended in the 1995 EPACMTP User's Guide (EPA 1995c). Appendix E describes the methodology used to calculate many of these input parameters.

- Because infiltration rates vary across the country due to changes in rainfall, soil type, etc., a distribution of infiltration rates were used corresponding to the climatic zones presented in the EPACMTP User's Guide. For the deterministic analyses, this distribution assumed that the non-utility locations were located uniformly across each of the climatic zones. This assumption is an estimation of the actual distribution of thousands of sites, but does not account for regional variances in location (e.g., a concentration of sites in more industrialized states or states more likely to burn coal).

4.5.2 Commercial Landfill (Scenario NL)

Most assumptions were the same as for the oil-fired waste commercial landfill scenario.

- EPA calculated unit dimensions from the same database of nonhazardous waste landfills used for the HWIR analysis. Not all landfills from this database were used in the analysis; only those landfills associated with certain FFC-waste generating industries were used. The median landfill area determined in this way was used with a representative average depth for all deterministic and Monte Carlo analyses in this scenario. These calculations are described in detail in Appendix G.

- EPA assumed that 54 percent of the wastes in the landfill were non-utility FFC wastes. This proportion was based on median waste generation rates and the commercial landfill size derived as noted above. Each generator was assumed to use a different landfill.

- EPA assumed the landfill was unlined.

- The model assumes that the same leachate characteristic data describe the leachate from the entire unit.

- EPA used the same waste concentration, meteorologic, and hydrogeologic parameters data as used in the monofill scenario. The receptor well distance, finite source assumption, and 10,000 year study period are the same as discussed for coal comanagement wastes.

Table 4-4. Summary of EPACMPT Model Inputs (see Appendix A for data sources and details)

CMTP Data Elements	Coal-fired utility comanagement			Oil-fired utility			FBC		Nonutiity	
	Surface Imp. [CS]	Landfill [CL]	Minefill [CF]	Surface Imp. [OS]	On-site monofill [OM]	Commercial landfill [OL]	Landfill [FL]	Minefill [FF]	On-site monofill [NM]	Commercial landfill [NL]
Source-specific variables										
AREA, management unit area (m^2)	CT: $0.364x10^6$ m^2 HE: $1.67x10^6$ m^2	CT: $0.267x10^6m^2$ HE: $1.33x10^6$ m^2	CT: 141,000 m^2	CT: 3,600 m^2 HE: 8,900 m^2	CT: 4,860 m^2	CT: 34,400 m^2	CT: $0.155x10^6$ m^2 HE: $0.317x10^6$ m^2	CT: 141,000 m^2	CT: 7,700 m^2 HE: 34,500 m^2	CT: 34,400 m^2
CZERO, leachate concentration	95th %ile of pore/leached concentration			95th %ile of extracted leachate values			CIBO Data		95th %ile of pore/leached concentration	
Cw/Cl value (waste to leachate concentration)	(n/a)	Constituent Dependent	Constituent Dependent	(n/a)	Constituent Dependent					
RECHRG, recharge rate	CT:0.3256	CT: 0.0894 m/y	CT: 0.0789 m/y	CT: 0.1016 m/y	CT: 0.1016 m/y	CT: 0.1016 m/y	CT: 0.0903 m/y	CT: 0.0789 m/y	CT: 0.1143 m/y HE: 0.0005 or 0.4384 m/y	CT: 0.1143 m/y HE: 0.0005 or 0.4384 m/y
SINFIL, infiltration rate from unit	Derived			Derived (m/y)						
DEPTH, depth of landfill	Derived	CT: 9.45 m HE: 33.53 m	CT: 7.56 m	(n/a)	CT: 3.89 m	CT: 2.25 m	CT: 15.8 m HE: 22.9 m	CT:7.56 m	CT: 5.3 m HE: 13.3 m	CT: 2.25 m
waste fraction	CT: 100%	CT: 100%	CT: 100%	CT: 100%	CT: 100%	CT: 24%	CT: 100%	CT: 100%	CT: 100%	CT: 56%
waste density	(n/a)	CT: 1.19 g/cm^3	CT: 1.19 g/cm^3	(n/a)	CT: 1.19 g/cm^3	CT: 1.19 g/cm^3	CT: 1.19 g/cm^3	CT:1.19g/cm^3	CT: 1.19 g/cm^3	CT: 1.19 g/cm^3
HZERO, ponding depth of surface impoundment , m	CT: 1.8 m HE: 19 m	(n/a)	(n/a)	CT: 1.17 m HE: 2.6 m	(n/a)	(n/a)	(n/a)	(n/a)	(n/a)	(n/a)
DLINR, liner thickness, m	CT: 3.4 m HE: 0.43 m	(n/a)	(n/a)	CT: 0.21 m HE: 0.098 m	(n/a)	(n/a)	(n/a)	(n/a)	(n/a)	(n/a)
CLINR, hydraulic conductivity of liner	CT: 0.315 m/y	(n/a)	(n/a)	CT: 0.315 m/y	(n/a)	(n/a)	(n/a)	(n/a)	(n/a)	(n/a)

Table 4-4. Summary of EPACMPT Model Inputs (see Appendix A for data sources and details)

CMTP Data Elements	Coal-fired utility comanagement			Oil-fired utility			FBC		Nonutiity	
	Surface Imp. [CS]	Landfill [CL]	Minefill [CF]	Surface Imp. [OS]	On-site monofill [OM]	Commercial landfill [OL]	Landfill [FL]	Minefill [FF]	On-site monofill [NM]	Commercial landfill [NL]
TSOURC, duration of leaching	Constant: 40 years	Up to 10,000 years	Up to 10,000 years	Constant: 40 yrs	Up to 10,000 years					
Metals-specific variables										
METAL_ID	Constituent dependent--only metals with iostherms									
USPH, Soil and aquifer pH	CT: 6.92 HE: 4.73 or 9.02	CT: 6.92 HE: 4.73 or 9.02	CT: 6.80	CT: 6.92 HE: 4.73 or 9.02	CT: 6.92 HE: 4.73 or 9.02	CT: 6.80	CT: 6.92 HE: 4.73 or 9.02	CT: 6.80	CT: 6.80	CT: 6.80
FEOX, iron hydroxide concentration in soil and aquifer	CT: 0.562% HE: 0.0675 or 1.057% (HWIR Distribution)									
LOM, concentration of dissolved organic carbon in the waste leachate	CT: 9.49 mg/L HE: 1.44 or 181 mg/L	CT: 9.49 mg/L HE: 1.44 or 181 mg/L	CT: 49.8 mg/L	CT: 9.49 mg/L HE: 1.44 or 181 mg/L	CT: 9.49 mg/L HE: 1.44 or 181 mg/L	CT: 49.8 mg/L	CT: 9.49 mg/L HE: 1.44 or 181 mg/L	CT: 49.8 mg/L	CT: 49.8 mg/L	CT: 49.8 mg/L
USNOM, unsaturated zone percentage organic matter (should be same as POM)	CT: 1.58 (%) HE: 0.35 or 4.50 (%)	CT: 1.58 (%) HE: 0.35 or 4.50 (%)	CT: 0.105 (%)	CT: 1.58 (%) HE: 0.35 or 4.50 (%)	CT: 1.58 (%) HE: 0.35 or 4.50 (%)	CT: 0.105 (%)	CT: 1.58 (%) HE: 0.35 or 4.50 (%)	CT: 0.105 (%)	CT: 0.105 (%)	CT: 0.105 (%)
ASNOM, aquifer fraction organic carbon (should be same as FOC)	CT: 0.032 HE: 0.061 or 0.003 (HWIR Default)									

Table 4-4. Summary of EPACMPT Model Inputs (see Appendix A for data sources and details)

CMTP Data Elements	Coal-fired utility comanagement			Oil-fired utility			FBC		Nonutiity	
	Surface Imp. [CS]	Landfill [CL]	Minefill [CF]	Surface Imp. [OS]	On-site monofill [OM]	Commercial landfill [OL]	Landfill [FL]	Minefill [FF]	On-site monofill [NM]	Commercial landfill [NL]
Unsaturated Zone Variables										
Saturated conductivity	Constant: 0.343 cm/hr (HWIR mean value for silt loam)									
α moisture retention parameter	Constant: 0.019 cm^{-1} (HWIR mean value for silt loam)									
β moisture retention parameter	Constant: 1.409 (HWIR mean value for silt loam)									
Res. Water content	Constant: 0.068 (HWIR mean value for silt loam)									
Sat. Water content	Constant: 0.45 (HWIR mean value for silt loam)									
DSOIL, thickness of unsaturated zone	CT: 8.3 m HE: 0 m	CT: 8.3 m HE: 0 m	CT: 6.1 m	CT: 6.98 m HE: 2.2 m	CT: 6.98 m HE: 2.2 m	CT: 4.65 m	CT: 8.3 m HE: 0 m	CT: 6.1 m	CT: 3.55 m	CT: 3.55 m
Dispersivity	Derived (HWIR)									
% organic matter	CT: 1.58 (%) HE: 0.35 or 4.50 (%)	CT: 1.58 (%) HE: 0.35 or 4.50 (%)	CT: 0.105 (%)	CT: 1.58 (%) HE: 0.35 or 4.50 (%)	CT: 1.58 (%) HE: 0.35 or 4.50 (%)	CT: 0.105 (%)	CT: 1.58 (%) HE: 0.35 or 4.50 (%)	CT: 0.105 (%)	CT: 0.105 (%)	CT: 0.105 (%)
Bulk density	CT: 1.42 g/cm^3 HE: 1.85 or 0.89 g/cm^3	CT: 1.42 g/cm^3 HE: 1.85 or 0.89 g/cm^3	CT: 1.65 g/cm^3	CT: 1.42 g/cm^3 HE: 1.85 or 0.89 g/cm^3	CT: 1.42 g/cm^3 HE: 1.85 or 0.89 g/cm^3	CT: 1.65 g/cm^3	CT: 1.42 g/cm^3 HE: 1.85 or 0.89 g/cm^3	CT: 1.65 g/cm^3	CT: 1.65 g/cm^3	CT: 1.65 g/cm^3
Saturated Zone Parameters										
DIAM, average particle diameter in aquifer	CT: 0.021 cm HE: 8.9e-04 or 0.23 cm (HWIR)									
POR, aquifer porosity	CT: 0.41 HE: 0.32 or 0.53 (HWIR Default)									

Table 4-4. Summary of EPACMPT Model Inputs (see Appendix A for data sources and details)

CMTP Data Elements	Coal-fired utility comanagement			Oil-fired utility			FBC		Nonutiity	
	Surface Imp. [CS]	Landfill [CL]	Minefill [CF]	Surface Imp. [OS]	On-site monofill [OM]	Commercial landfill [OL]	Landfill [FL]	Minefill [FF]	On-site monofill [NM]	Commercial landfill [NL]
BULKD, aquifer bulk density	CT: 1.56 g/cm^3 HE: 1.25 or 1.80 (HWIR Default)									
ZB, aquifer saturated thickness	CT: 15.20 m			CT: 15.20 m			CT: 7.62 m	CT: 15.20 m	CT: 7.09 m	CT: 7.09 m
XKX, longitudinal hydraulic conductivity, K_x	CT: 315 m/y	CT: 315 m/y	CT: 300 m/y	CT: 315 m/y			CT: 631 m/y	CT: 300 m/y	CT: 473 m/y	CT: 473 m/y
Anistropy ratio	1 (Assumed)									
GRADNT, hydraulic gradient	CT: 0.009	CT: 0.009	CT: 0.009	CT: 0.009	CT: 0.009	CT: 0.009	CT: 0.005	CT: 0.009	CT: 0.005	CT: 0.005
VXCS, regional groundwater seepage velocity	Derived (m/y)									
AL, longitudinal dispersivity	CT: 4.64 m HE: 0.32 or 68 m (HWIR Distribution)									
AT, transverse dispersivity ratio	Constant: 8 (HWIR Recommendation)									
AV, vertical dispersivity ratio	Constant: 160 (HWIR Recommendation)									
TEMP, temperature of ambient aquifer water	CT: 17.5 C	CT: 12.5 C	CT: 12.5 C	CT: 22.5 C	CT: 22.5 C	CT: 22.5 C	CT: 15 C	CT: 12.5 C	CT: 12.5 C	CT: 12.5 C
PH, ambient groundwater pH	CT: 6.92 HE: 4.73or 9.02	CT: 6.92 HE: 4.73or 9.02	CT: 6.80	CT: 6.92 HE: 4.73or 9.02	CT: 6.92 HE: 4.73or 9.02	CT: 6.80	CT: 6.92 HE: 4.73or 9.02	CT: 6.80	CT: 6.80	CT: 6.80

Table 4-4. Summary of EPACMPT Model Inputs (see Appendix A for data sources and details)

CMTP Data Elements	Coal-fired utility comanagement			Oil-fired utility			FBC		Nonutiity	
	Surface Imp. [CS]	Landfill [CL]	Minefill [CF]	Surface Imp. [OS]	On-site monofill [OM]	Commercial landfill [OL]	Landfill [FL]	Minefill [FF]	On-site monofill [NM]	Commercial landfill [NL]
FOC, fraction organic carbon	CT: 0.032 HE: 0.061 or 0.003 (HWIR Default)									
Receptor Well Location	HE: 150 meters, on centerline, depth is at water table (Assumed)									

Notes
1. CT= “Central Tendency”
2. HE= “High End”
3. Appendix A contains data sources for all input parameters.

5. MODELING RESULTS

5.1 OVERVIEW

This section presents the screening, high-end deterministic modeling, and probabilistic modeling results obtained for all of the scenarios detailed in the preceding chapter. For simplicity, the screening, high-end, and probabilistic results for all scenarios for each remaining FFC waste category are grouped into separate subsections. Each subsection ends with a brief discussion of the results for the given remaining waste category. The following paragraphs describe the manner in which results are presented.

Screening results are presented in table format, listing the constituent of concern, the human health benchmark (HBL) derived for the constituent, the high-end (95th percentile) concentration derived for the constituent, and the screening result.[1] The screening result is defined as the ratio of the high-end (95^{th} rank-ordered percentile) constituent leachate concentration (C_L) to the HBL.

$$\text{Screening Result} = C_L/\text{HBL}$$

EPA defined the screening threshold for this exercise to be a value of 1. For any constituent for which the screening result is less than 1, EPA eliminated the constituent from further consideration. All constituents for which the 95th percentile concentration exceeded the HBL were retained for deterministic and probabilistic modeling. For ease of presentation, all screening results exceeding a value of 1 are shown in bold italics.

As discussed in Section 4, all scenarios for a given FFC waste category were ascribed the same waste characteristics. For example, the concentration data for the coal-fired utility comanaged waste surface impoundment scenario were identical to the data used for the coal-fired utility

[1]The screening tables in this section present results only for those constituents that can be effectively handled by EPACMTP. As discussed in Section 2, screening was conducted for a larger set of constituents, some of which lack the requisite data to be run in EPACTMP. Complete screening tables are included in Appendix C.

comanaged waste minefill scenario.[2] Accordingly, any constituent that exceeded the HBL was examined for all scenarios appropriate for that remaining waste category.

High-end deterministic modeling results are presented in a manner similar to the screening results. Tables for each scenario include the constituent of concern, the associated benchmark value (HBL), the high-end leachate concentration, and the predicted dilution and attenuation factor (DAF) from the model and the corresponding estimate of risk, expressed as a hazard quotient (HQ) or individual lifetime cancer risk. The DAF is defined as the initial leachate concentration (C_L) divided by the predicted peak concentration (C_P) in the down-gradient receptor well:

$$DAF = C_L/C_P.$$

For the high-end deterministic assessment, C_L was defined to be the 95th percentile leachate concentration, and the location of the monitoring well for which C_P is predicted is the high-end monitoring well distance of 150 meters. The DAF reflects the magnitude of the dilution and attenuation affecting the constituents of concern released from the modeled waste management unit as the contaminant migrate toward the receptor well.[3]

Risk is presented in the results tables as a Hazard Quotient (HQ)[4] or the individual lifetime cancer risk (for carcinogens only).[5] The HQ was defined to be the ratio of the peak receptor well concentration (C_P) to the HBL:

[2] Limitations to this assumption are discussed in detail in Section 7.

[3] The larger the DAF, the smaller the predicted peak down-gradient concentration, and therefore the greater the dilution and attenuation affecting the fate and transport of the constituent. Accordingly, a DAF of 1 reflects conditions of zero dilution or attenuation. An infinite DAF indicates that no migration has occurred at all.

[4] As defined, the HQ provides an estimate of the potential risk to the individual receptor from exposure to contaminated ground water characterized by the concentration C_P: the higher the receptor well concentration relative to the HBL, the higher the risk to the receptor exposed. As with the screening assessment, EPA defined the risk threshold to be an HQ of 1, such that all constituents modeled for a given scenario resulting in an HQ less than 1 are expected to present very low risk. Alternatively, waste constituents showing an HQ much greater than 1 for a given scenario require additional consideration as potentially presenting unacceptable risk.

[5] For carcinogens, As and Be, the corresponding equation for lifetime individual cancer risk is given as Risk = $(C_P/HBL)\times10^{-6}$. The corresponding threshold value is 1×10^{-6}.

$$HQ = C_p/HBL.$$

Taken together, the DAF and the HQ provide a two-sided description of the deterministic model result: the DAF indicates the extent to which a constituent of concern may migrate to the receptor well, and the HQ indicates the risk that the resulting concentration may present to the hypothetical receptor. The high-end results require further analysis, however, both to determine that the scenario provides a true high-end estimate of risk, and to determine the effects of modeling uncertainty on the model outcomes. EPA performed probabilistic analysis to determine that the scenarios provided high-end results, and the results are discussed below. EPA's uncertainty analysis is presented in Section 7.

The purpose of the Monte Carlo analysis is to verify that the deterministic analysis represents a true high-end evaluation of risk. Conceptually, "high end" exposure means exposure above the 90th percentile of the population (EPA 1995b). Therefore, if the Monte Carlo analysis shows that the deterministic high end analysis is at or above the 90th percentile, the parameters selected for high end analysis were adequate. Similarly, results below the 90th percentile reveal that further investigation is required. Such a result would demonstrate one of two conclusions: (1) there may have been other parameters that would have been better suited to represent the high-end scenario, or (2) the model is unstable for these constituents around the given concentrations, and give higher than expected risks in the Monte Carlo framework.

As discussed previously, the Monte Carlo analysis for each constituent for each scenario was executed by running a series of 2000 deterministic model runs and ranking the results to determine where in the output distribution the high-end scenario fell. Each of the 2000 iterations was defined by EPACMTP by allowing all of the model input values to be selected at random from the appropriate input value distributions. Based on the output, EPA sought to answer two primary questions: do the high-end modeling results fall in the upper 10 percent of all Monte Carlo output values, and do the high-end Monte Carlo results corroborate the findings of the high-end analysis with respect to risk.

Monte Carlo results are presented in table format showing the constituent of concern, the 50th percentile and 95th percentile Monte Carlo results, the high-end result for the same waste management scenario (transcribed from the earlier tables), and the percentile ranking of the high-end result within the Monte Carlo results distribution.[6]

5.2 COAL-FIRED UTILITY COMANAGED WASTES

5.2.1 Screening Analysis

Table 5-1 shows screening analysis results for coal-fired utility comanaged wastes. The screening analysis was conducted as described in the preceding sections of this report. No characterization data were available for these wastes for thallium. Therefore, screening was not conducted for thallium. Antimony and silver were not detected in any of the coal-fired utility comanaged waste samples. Copper, barium, and mercury were detected, but at 95th percentile concentrations below the screening criteria. Accordingly, these constituents were eliminated from further consideration. All other EPACMTP constituents survived for analysis in modeling.

5.2.2 High-end Analysis

High-end deterministic modeling was conducted for all surviving EPACMTP constituents for coal-fired utility comanaged wastes using three distinct waste management scenarios: surface impoundment, a landfill, and a minefill. This modeling was conducted as described in the preceding sections of this report. The results for each scenario are presented individually below.

[6] To clarify, if the high-end scenario yielded an HQ = 7, and if 1,900 of the of the 2,000 Monte Carlo simulations were less than 7, the high-end HQ would correspond to the 95th percentile result in the Monte Carlo distribution. In this case, the Monte Carlo result would confirm that the high-end scenario yielded a true high-end estimate of the potential risk for that constituent. Alternatively, if the high-end HQ = 3 and only 1,600 of the Monte Carlo simulations yielded results less than 3, the high-end HQ would correspond to the 80th percentile result in the Monte Carlo distribution. In this case, the Monte Carlo result would indicate that the high-end scenario as defined did not yield a true high-end result. For that constituent, for that scenario, other selections of high-end parameters would yield a more conservative estimate of risk, all else being equal.

Surface Impoundment (Scenario CS)

Table 5-1. Screening Results for Coal-fired Utility Comanaged Wastes

Constituent	HBL[1] (mg/l)	95th % Observed Concentration (mg/l)	Screening Result and Conclusion[2]
Antimony	0.021	Not detected	0
Arsenic	0.00029-(c)	9.64	***33,241***
Barium	3.60	1.04	0.29
Cadmium	0.026	0.156	***6.00***
Chromium III/VI	0.26	0.746	***2.87***
Copper	1.3-(a)	0.69	0.53
Lead	0.015-(a)	0.468	***31.2***
Mercury	0.015	0.000796	0.053
Nickel	1.03	8.33	***8.09***
Selenium	0.257	1.03	***4.01***
Silver	0.257	Not detected	0
Vanadium	0.36	0.8	***2.22***
Zinc	15.4	23.1	***1.50***

Notes:
1. All HBLs listed above are non-carcinogens, except for arsenic -(c); lead and copper are action levels -(a), not health based numbers.
2. Numbers in **bold** and *italics* indicate which constituents exceed the screening criteria.

Table 5-2 shows high-end analysis results for the coal-fired utility comanaged waste surface impoundment (Scenario CS). Arsenic is the only constituent that exceeded the risk threshold under the assumptions used for this scenario. Arsenic exceeded the threshold by a factor of approximately 500. For a few constituents (chromium III, lead, and vanadium), predicted receptor well concentrations were near or equal to zero at all times within the modeling period (10,000 years). For the other constituents, predicted peak receptor well concentrations ranged from approximately 1/3 to 1/100 of the risk threshold. These results are discussed in greater detail in Section 5.2.4.

Landfill (Scenario CL)

Table 5-3 shows high-end analysis results for the coal-fired utility comanaged waste landfill (Scenario CL). Several constituents exceeded the risk threshold under the assumptions used for this scenario: arsenic, chromium VI, nickel, and selenium. Arsenic exceeded the threshold by more than four orders of magnitude. Nickel was approximately four times the threshold, while exceedences for

Table 5-2. Results for Coal-fired Utility Comanaged Waste Surface Impoundment (Scenario CS), High-end Analysis

Constituent	HBL[1] (mg/l)	95th % Observed Conc. (mg/l)	DAF[2] Result from Modeling	HQ and Conclusion[3]
Arsenic	0.00029-(c)	9.64	65.4	***Risk = 5.08×10^{-4}***
Cadmium	0.026	0.156	215	0.028
Chromium III	0.26	0.746	$>10^{6}$	$<10^{-6}$
Chromium VI			41.3	0.069
Lead[4]	0.015	0.468	$>10^{18}$	$<10^{-18}$
Nickel	1.03	8.33	95.4	0.085
Selenium	0.257	1.03	10.9	0.37
Vanadium	0.36	0.8	$>10^{19}$	$<10^{-19}$
Zinc	15.4	23.1	117	0.013

Notes:
1. All HBLs are non-carcinogens, except for arsenic -(c).
2. DAF result is the ratio of the initial concentration to the predicted receptor well peak concentration.
3. HQ is predicted receptor well peak concentration divided by HBL.
4. Lead is an action level, not a health based number.
5. Numbers in **bold** and *italics* indicate which constituents exceed the risk threshold (HQ = 1 or risk = 10^{-6}).

chromium VI and selenium were each less than twice the threshold. Predicted receptor well concentrations for lead and vanadium were near or equal to zero at all times within the modeling period (10,000 years). Predicted peak receptor well concentrations for cadmium and zinc also were small compared to their HBLs. These results are discussed in greater detail in Section 5.2.4.

Minefill (Scenario CF)

Table 5-4 shows high-end analysis results for the coal-fired utility comanaged waste minefill (Scenario CF). Constituents exceeding the risk threshold under the assumptions used for this scenario are the same as those in the landfill scenario: arsenic, chromium VI, nickel, and selenium. The magnitude of each exceedence was slightly smaller for this scenario (just under four orders of magnitude for arsenic, under four times for nickel, and 28 and 42 percent for chromium VI and selenium, respectively). Of the constituents that did not exceed the threshold, only zinc came close at 68 percent of its HBL. Predicted receptor well concentrations for lead and vanadium were near or equal to zero at all times within the modeling period (10,000 years). These results are discussed in greater detail in Section 5.2.4.

Table 5-3. Results for Coal-fired Utility Comanaged Waste Landfill (Scenario CL), High-end Analysis

Constituent	HBL[1] (mg/l)	95th % Observed Conc. (mg/l)	DAF[2] Result from Modeling	HQ and Conclusion[3,5]
Arsenic	0.00029-(c)	9.64	3.17	***Risk = 1.05×10^{-2}***
Cadmium	0.026	0.156	23,708	0.00025
Chromium III	0.26	0.746	$>10^{6}$	$<10^{-5}$
Chromium VI			1.84	***1.56***
Lead[4]	0.015	0.468	$>10^{18}$	$<10^{-17}$
Nickel	1.03	8.33	1.77	***4.57***
Selenium	0.257	1.03	2.25	***1.78***
Vanadium	0.36	0.8	$>10^{12}$	0.000000000001
Zinc	15.4	23.1	$>10^{7}$	0.000000088

Notes:
1. All HBLs are non-carcinogens, except for arsenic -(c).
2. DAF result is the ratio of the initial concentration to the receptor well peak concentration.
3. HQ is predicted receptor well peak concentration divided by HBL.
4. Lead is an action level, not a health based number.
5. Numbers in **bold** and *italics* indicate which constituents exceed the risk threshold (HQ = 1 or risk = 10^{-6}).

5.2.3 Central Tendency (Monte Carlo) Analysis

Central tendency analysis was conducted using probabilistic Monte Carlo modeling for two of the three coal-fired utility comanaged waste scenarios (surface impoundment and landfill) for all surviving EPACMTP constituents. This modeling was conducted as described in the preceding sections of this report. The results for each of these two scenarios are presented individually below.Surface Impoundment (Scenario CS)

Table 5-5 presents the probabilistic risk results for the coal-fired utility comanaged waste surface impoundment (Scenario CS) and compares them to the deterministic results. At the 50th percentile, Monte Carlo modeling predicted only one constituent above the risk threshold: arsenic. At the 95th percentile, selenium also exceeded the threshold, by 16 percent. All other constituents remained well below the risk threshold even at the 95th percentile of the probabilistic results.

The deterministic high-end results for arsenic and zinc correspond to the 90th or greater percentile of the Monte Carlo results. The deterministic results for the other constituents fall

Table 5-4. Results for Coal-fired Utility Comanaged Waste Minefill (Scenario CF), High-end Analysis

Constituent	HBL[1] (mg/l)	95th % Observed Conc. (mg/l)	DAF[2] Result from Modeling	HQ and Conclusion[3,5]
Arsenic	0.00029-(c)	9.64	3.90	***Risk = 8.52×10^{-3}***
Cadmium	0.026	0.156	26,691	0.00022
Chromium III	0.26	0.746	$>10^{6}$	$<10^{-6}$
Chromium VI			2.25	***1.28***
Lead[4]	0.015	0.468	$>10^{18}$	$<10^{-18}$
Nickel	1.03	8.33	2.29	***3.53***
Selenium	0.257	1.03	2.83	***1.42***
Vanadium	0.36	0.8	$>10^{9}$	$<10^{-10}$
Zinc	15.4	23.1	2.18	0.69

Notes:
1. All HBLs are non-carcinogens, except for arsenic -(c).
2. DAF result is the ratio of the initial concentration to the receptor well peak concentration.
3. HQ is predicted receptor well peak concentration divided by HBL.
4. Lead is an action level, not a health based number.
5. Numbers in **bold** and *italics* indicate which constituents exceed the risk threshold (HQ = 1 or risk = 10^{-6}).

below the 90th percentile, but were still toward the upper end (except for lead and cadmium, whose concentrations were infinitesimal and whose percentile results, therefore, are not meaningful). These results are discussed in greater detail in Section 5.2.4.

Landfill (Scenario CL)

Table 5-6 presents the probabilistic risk results for the coal-fired utility comanaged waste landfill (Scenario CL) and compares them to the deterministic results. At both the 50th and 95th percentiles, Monte Carlo modeling predicted only one constituent above the risk threshold: arsenic. All other constituents remained below the risk threshold even at the 95th percentile of the probabilistic results.

The deterministic high-end results for four constituents correspond to the 90th or greater percentile of the Monte Carlo results; each of these deterministic results fall in the 99th percentile. Only cadmium, lead, vanadium, and zinc fell below the 90th percentile. For all of these,

Table 5-5. Comparison of Probabilistic and Deterministic Risk Results for Coal-fired Utility Comanaged Waste Surface Impoundment (Scenario CS)

Constituent	Predicted HQ[1] or Risk			
	Monte Carlo		High-end Analysis	
	50th percentile[2]	95th percentile[2]	Result[2]	Corresponding Monte Carlo Percentile
Arsenic (risk)	***1.03×10⁻⁵***	***1.38×10⁻³***	***5.08⁻²×10⁻⁴***	90th
Cadmium	0.00114	0.137	0.028	84th
Chromium VI	0.00701	0.41	0.069	79th
Lead	0	0.0306	$<10^{-18}$	55th
Nickel	0.0017	0.193	0.085	89th
Selenium	0.0443	***1.16***	0.37	81th
Vanadium	0	0.14	$<10^{-19}$	76th
Zinc	0.0000307	0.00646	0.013	97th

Notes:
1. Values shown are HQs except for arsenic and beryllium, which are risk. HQ is the predicted well concentration divided by the HBL.
2. Numbers in **bold** and *italics* indicate which constituents exceed the risk threshold (HQ=1 or risk=10^{-6}) in a given percentage of model runs.

predicted receptor well concentrations were near or equal to zero for nearly all of the Monte Carlo runs (as they were in the deterministic case). These results are discussed in greater detail in Section 5.2.4.

5.2.4 Discussion of Results

The screening and modeling results for the comanaged waste scenarios indicate that, while most of the trace constituents of concern in comanaged wastes exceeded the screening level concentrations, few of these constituents were predicted to exceed the same benchmark values in near-by down-gradient ground water. Arsenic was predicted to exceed the benchmark values in ground water for each of the three scenarios, with the calculated risk ranging from 5×10^{-4} to 1×10^{-2} increased individual lifetime risk of cancer. For the landfill and minefill scenarios, several additional constituents exceeded their respective thresholds by less than a factor of 10.

The Monte Carlo simulation results strongly corroborated the conservatism of the high-end analyses for comanaged wastes. For all constituents found to exceed the benchmark value in the

Table 5-6. Comparison of Probabilistic and Deterministic Risk Results for Coal-fired Utility Comanaged Waste Landfill (Scenario CL)

Constituent	Predicted HQ[1] or Risk			
	Monte Carlo		High-end Analysis	
	50th percentile[2]	95th percentile[2]	Result[2]	Corresponding Monte Carlo Percentile
Arsenic (risk)	***1.87×10⁻⁶***	***1.11×10⁻³***	***1.05×10⁻²***	99th
Cadmium	0.0000000001	0.060	0.00025	75th
Chromium VI	0.0019	0.36	***1.56***	99th
Lead	0	0.00015	$<10^{-17}$	83rd
Nickel	0.00000032	0.092	***4.57***	99th
Selenium	0.0087	0.46	***1.78***	99th
Vanadium	0	0.075	0.000000000001	77th
Zinc	0.000031	0.0065	0.000000088	22nd

Notes:
1. Values shown are HQs except for arsenic and beryllium, which are risk. HQ is the predicted well concentration divided by the HBL.
2. Numbers in **bold** and *italics* indicate which constituents exceed the risk threshold (HQ=1 or risk=10^{-6}) in a given percentage of model runs.

deterministic analyses, the high-end predicted concentration equaled or exceeded the corresponding 90th percentile concentration predicted in the Monte Carlo simulation. For example, the arsenic, chromium (VI), nickel and selenium, high-end risk levels for the landfill scenario corresponded to the 99th percentile Monte Carlo result. In fact, the 95th percentile Monte Carlo risk estimation fell below the threshold value for all metals in all scenarios except for arsenic and selenium (selenium in the impoundment only).

The high-end deterministic analyses generally predicted higher risk in conjunction with the landfill and minefill compared with the surface impoundment. Similarly, the high-end results demonstrated slightly higher risks for the landfill than the minefill. Comparing the Monte Carlo results for the impoundment and the landfill scenarios, however, indicates that the risks predicted for the surface impoundment were similar to those predicted for the landfill, at both the 50th and the 95th percentile level.

The generally low level of exceedence for nickel and selenium in the landfill and minefill scenarios, supported with the probabilistic results all falling below threshold risk levels at the 95th percentile level, suggest low expected risk from these constituents. Additionally, as explained in Appendix H, the chromium risks predicted were based on a very conservative overestimate of hexavalent chrome in leachate, and so are not significant.

Overall, the comanaged waste scenarios indicate that arsenic remains a constituent of concern for all scenarios. Please refer to Section 7 for a detailed discussion of the uncertainties relating to the quantitative estimate of risk.

5.3 OIL-FIRED UTILITY WASTES

5.3.1 Screening Analysis

Table 5-7. Screening Results for Oil-fired Utility Comanaged Wastes

Constituent	HBL[1] (mg/l)	95th % Observed Concentration (mg/l)	Screening Result and Conclusion[2]
Arsenic	0.00029-(c)	4.15	***14,310***
Barium	3.6	12.9	***3.58***
Cadmium	0.026	0.62	***23.9***
Chromium III/IV	0.26	3.44	***13.2***
Copper	1.3-(a)	3.415	***2.63***
Lead	0.015-(a)	13.4	***893***
Mercury	0.015	0.5	**33.3**
Nickel	1.03	470	***456***
Selenium	0.257	0.37	***1.44***
Silver	0.257	0.15	0.58
Vanadium	0.36	882	***2,450***
Zinc	15.4	8.12	0.53

Notes:
1. All HBLs listed above are non-carcinogens, except for arsenic -(c); lead and copper are action levels -(a), not health based numbers.
2. Numbers in **bold** and *italics* indicate which constituents exceed the screening criteria.

Table 5-7 shows screening analysis results for oil-fired utility wastes. The screening analysis was conducted as described in the preceding sections of this report. No characterization data were available for antimony or thallium. Therefore, screening was not conducted for these constituents.

Table 5-8. Results for Oil-fired Utility Waste Surface Impoundment (Scenario OS), High-end Analysis

Constituent	HBL[1] (mg/l)	95th % Observed Conc. (mg/l)	DAF[2] Result from Modeling	HQ and Conclusion[3,5]
Arsenic	0.00029-(c)	4.15	56.0	***Risk = 2.56×10^{-4}***
Barium	3.60	12.9	20.9	0.17
Cadmium	0.026	0.62	89.8	0.27
Chromium III	0.26	3.44	>10^{6}	<0.0001
Chromium VI			35.3	0.37
Copper	1.3	3.415	>10^{20}	<10^{-20}
Lead[4]	0.015	13.4	>10^{22}	<10^{-20}
Mercury	0.015	0.5	137	0.24
Nickel	1.03	470	4.51	***101***
Selenium	0.257	0.37	9.33	0.15
Vanadium	0.36	882	2.59	***946***

Notes:
1. All HBLs listed above are non-carcinogens, except for arsenic -(c).
2. DAF result is the ratio of the initial concentration to the receptor well peak concentration.
3. HQ is predicted receptor well peak concentration divided by HBL.
4. Lead is an action level, not a health based number.
5. Numbers in **bold** and *italics* indicate which constituents exceed the risk threshold (HQ = 1 or risk = 10^{-6}).

The 95th percentile concentrations for silver and zinc were below the screening criteria. Accordingly, these constituents were eliminated from further consideration. All other EPACMTP constituents survived for analysis in modeling.

5.3.2 High-end Analysis

High-end deterministic modeling was conducted for all surviving EPACMTP constituents for oil-fired utility comanaged wastes using three distinct waste management scenarios: a surface impoundment, an onsite monofill, and an offsite commercial landfill. This modeling was conducted as described in the preceding sections of this report. The results for each scenario are presented individually below.

Table 5-9. Results for Oil-fired Utility Waste Onsite Monofill (Scenario OM), High-end Analysis

Constituent	HBL[1] (mg/l)	95th % Observed Conc. (mg/l)	DAF[2] Result from Modeling	HQ and Conclusion[3,5]
Arsenic	0.00029-(c)	4.15	158	***Risk = 9.1×10^{-5}***
Barium	3.60	12.9	147	0.024
Cadmium	0.026	0.62	227	0.11
Chromium III	0.26	3.44	>10^6	<0.0001
Chromium VI			155	0.085
Copper	1.3	3.415	2,139	0.0012
Lead[4]	0.015	13.4	>10^6	<0.001
Mercury	0.015	0.5	>10^6	<0.0001
Nickel	1.03	470	133	***3.43***
Selenium	0.257	0.37	145	0.0099
Vanadium	0.360	882	294	***8.33***

Notes:
1. All HBLs listed above are non-carcinogens, except for arsenic -(c).
2. DAF result is the ratio of the initial concentration to the receptor well peak concentration.
3. HQ is predicted receptor well peak concentration divided by HBL.
4. Lead is an action level, not a health based number.
5. Numbers in **bold** and *italics* indicate which constituents exceed the risk threshold (HQ =1).

Surface Impoundment (Scenario OS)

Table 5-8 shows high-end analysis results for the oil-fired utility waste surface impoundment (Scenario OS). Arsenic, nickel, and vanadium exceeded the risk threshold under the assumptions used for this scenario. Vanadium exceeded the threshold by a factor of nearly 1,000, arsenic by a factor of over 200, and nickel by a factor of 100. For a few constituents (chromium III, copper, and lead), predicted receptor well concentrations were near or equal to zero at all times within the modeling period (10,000 years). For the other constituents, predicted peak receptor well concentrations were less than or equal to approximately 1/3 of the risk threshold. These results are discussed in greater detail in Section 5.3.4.

Onsite Monofill (Scenario OM)

Table 5-9 shows high-end analysis results for the oil-fired utility monofill (Scenario OM). As they did for the oil-fired utility impoundment, arsenic, nickel, and vanadium exceeded the

riskthreshold in this scenario. The magnitude of exceedences, however, was smaller in this scenario(100 times for arsenic, eight times for vanadium, and three times for nickel). Predicted receptor well concentrations for chromium III, lead, and mercury were near or equal to zero at all times within the modeling period (10,000 years). Predicted peak receptor well concentrations for all other constituents also were small compared with their HBLs. These results are discussed in greater detail in Section 5.3.4.

Offsite Commercial Landfill (Scenario OL)

Table 5-10. Results for Oil-fired Utility Waste Offsite Commercial Landfill (Scenario OL), High-end Analysis

Constituent	HBL[1] (mg/l)	95th % Observed Conc. (mg/l)	DAF[2] Result from Modeling	HQ and Conclusion[3,5]
Arsenic	0.00029-(c)	4.15	6.04	***Risk = 2.37×10^{-3}***
Barium	3.60	12.9	3.38	***1.06***
Cadmium	0.026	0.62	4.7	***5.24***
Chromium III	0.26	3.44	>10^{6}	<0.0001
Chromium VI			5.38	***2.46***
Copper	1.3	3.415	3.37	0.78
Lead[4]	0.015	13.4	121,181	0.0074
Mercury	0.015	0.5	44,000	0.00076
Nickel	1.03	470	3.37	***135***
Selenium	0.257	0.37	3.58	0.40
Vanadium	0.36	882	6.74	***364***

Notes:
1. All HBLs listed above are non-carcinogens, except for arsenic -(c).
2. DAF result is the ratio of the initial concentration to the receptor well peak concentration.
3. HQ is predicted receptor well peak concentration divided by HBL.
4. Lead is an action level, not a health based number.
5. Numbers in **bold** and *italics* indicate which constituents exceed the risk threshold (HQ = 1 or risk = 10^{-6}).

Table 5-10 shows high-end analysis results for the oil-fired utility waste commercial landfill (Scenario OL). As in the other oil-fired utility scenarios, arsenic, nickel, and vandadium exceeded the risk threshold. Barium, cadmium, and chromium VI also exceeded the risk threshold in this scenario, although not by as great a degree as the other three constituents (2,000 times, 100 times, and 300 times for arsenic, nickel, and vanadium compared with 6 percent, five times, and two times

for barium, cadmium, and chromium VI). All other constituents were below the risk threshold, with only copper and selenium approaching the threshold (78 and 40 percent of the threshold, respectively). These results are discussed in greater detail in Section 5.3.4.

5.3.3 Central Tendency (Monte Carlo) Analysis

Central tendency analysis was conducted using probabilistic Monte Carlo modeling for each of the three oil-fired utility waste scenarios (surface impoundment, monofill, and commercial landfill) for all surviving EPACMTP constituents. This modeling was conducted as described in the preceding sections of this report. The results for each of these scenarios are presented individually below.

Surface Impoundment (Scenario OS)

Table 5-11 presents the probabilistic risk results for the oil-fired utility waste surface impoundment (Scenario OS) and compares them to the deterministic results. [Note that modeling is not complete for arsenic or selenium.] At the 50th percentile, Monte Carlo modeling predicted only one constituent above the risk threshold: vanadium. At the 95th percentile, nickel also exceeded the threshold. All other constituents remained below the risk threshold even at the 95th percentile of the probabilistic results.

The deterministic high-end results for all constituents except copper and lead correspond to the 90th or greater percentile of the Monte Carlo results. For these two exceptions, however, predicted receptor well concentrations were near or equal to zero for nearly all of the Monte Carlo runs (as they were in the deterministic case), so the percentile rank of the deterministic results for these two constituents is not meaningful. These results are discussed in greater detail in Section 5.3.4.

Onsite Monofill (Scenario OM)

Table 5-12 presents the probabilistic risk results for the oil-fired utility waste monofill (Scenario OM) and compares them to the deterministic results. At the 50th percentile, Monte Carlo modeling predicted only one constituent above the risk threshold: vanadium. At the 95th percentile,

Table 5-11. Comparison of Probabilistic and Deterministic Risk Results for Oil-fired Utility Waste Surface Impoundment (Scenario OS)

Constituent	Predicted HQ[1] or Risk			
	Monte Carlo		High-end Analysis	
	50th percentile[2]	95th percentile[2]	Result[2]	Corresponding Monte Carlo Percentile
Arsenic (risk)	[modeling not complete]		***Risk = 2.56×10^{-4}***	
Barium	0.00096	0.12	0.17	96th
Cadmium	0.00056	0.29	0.27	95th
Chromium	0.00084	0.052	0.37	99th
Copper	0.00000	0.011	<10^{-20}	85th
Lead	0.00000	0.035	<10^{-20}	69th
Mercury	0.000023	0.042	0.24	98th
Nickel	0.011	***2.41***	***101***	99th
Selenium	[modeling not complete]		0.15	
Vanadium	***11.20***	***384.17***	***946***	99th

Notes:
1. Values shown are HQs except for arsenic and beryllium, which are risk. HQ is the predicted well concentration divided by the HBL.
2. Numbers in **bold** and *italics* indicate which constituents exceed the risk threshold (HQ=1 or risk=10^{-6}) in a given percentage of model runs.

arsenic also exceeded the threshold. All other constituents remained below the risk threshold at the 95th percentile of the probabilistic results, although nickel was only slightly below at 94 percent.

The deterministic high-end results for most constituents correspond to the 90th or greater percentile of the Monte Carlo results. Copper, lead, mercury, and vanadium fall below the 90th percentile. For lead and mercury, predicted receptor well concentrations were near or equal to zero for nearly all of the Monte Carlo runs (as they were in the deterministic case). Therefore, the percentile rank of the deterministic results for these two constituents is not meaningful. These results are discussed in greater detail in Section 5.3.4.

Table 5-12. Comparison of Probabilistic and Deterministic Risk Results for Oil-fired Utility Waste Onsite Monofill (Scenario OM)

Constituent	Predicted HQ[1] or Risk			
	Monte Carlo		High-end Analysis	
	50th percentile[2]	95th percentile[2]	Result[2]	Corresponding Monte Carlo Percentile
Arsenic (risk)	1.93×10^{-7}	***1.15×10^{-4}***	***9.1×10^{-5}***	94th
Barium	0.00014	0.036	0.024	94th
Cadmium	0.00000119	0.155	0.11	94th
Chromium VI	0.00053	0.053	0.085	97th
Copper	0	0.011	0.0012	88th
Lead	0	0.012	<0.001	74th
Mercury	0	0.00024	<0.0001	54th
Nickel	0.00436	0.939	***3.43***	98th
Selenium	0.00016	0.0124	0.0099	94th
Vanadium	***1.23***	***64.6***	***8.33***	76th

Notes:
1. Values shown are HQs except for arsenic and beryllium, which are risk. HQ is the predicted well concentration divided by the HBL.
2. Numbers in **bold** and *italics* indicate which constituents exceed the risk threshold (HQ=1 or risk=10^{-6}) in a given percentage of model runs.

Table 5-13 presents the probabilistic risk results for the oil-fired utility waste commercial landfill (Scenario OL) and compares them to the deterministic results. At the 50th percentile, Monte Carlo modeling predicted all constituents well below the risk threshold. At the 95th percentile, only one constituent exceeded the risk threshold: arsenic.

The deterministic high-end results for most constituents correspond to the 90th or greater percentile of the Monte Carlo results. The majority of these deterministic results are in the 99th or greater percentile. Only mercury falls below the 90th percentile. These results are discussed in greater detail in Section 5.3.4.

5.3.4 Discussion of Results

Table 5-13. Comparison of Probabilistic and Deterministic Risk Results for Oil-fired Utility Waste Offsite Commercial Landfill (Scenario OL)

	Predicted HQ or Risk			
	Monte Carlo		High-end Analysis	
Constituent	50th percentile	95th percentile	Result	Corresponding Monte Carlo Percentile
Arsenic (risk)	4.40×10^{-7}	***2.42×10^{-4}***	***2.37×10^{-3}***	99th
Barium	0.00092	0.125	***1.06***	>100th
Cadmium	0.0000098	0.413	***5.24***	99th
Chromium VI	0.00132	0.0928	***2.46***	>100th
Copper	0	0.05733	0.78	>100th
Lead	0	0.0244	0.0074	94th
Mercury	0.00000022	0.007527	0.00076	84th
Nickel	0.0000000031	0.00011	***135***	>100th
Selenium	0.00056	0.03616	0.40	>100th
Vanadium	0.0004	0.02581	***364.0***	>100th

Notes:
1. Values shown are HQs except for arsenic and beryllium, which are risk. HQ is the predicted well concentration divided by the HBL.
2. Numbers in **bold** and *italics* indicate which constituents exceed the risk threshold (HQ=1 or risk=10^{-6}) in a given percentage of model runs.

The screening and modeling results for oil-fired utility wastes indicated that, despite generally high concentrations of metals of concern in the wastes, few metals were predicted to exceed benchmark levels in down-gradient ground-water receptor wells. Arsenic was predicted to exceed benchmark levels in all three oil ash management scenarios, with calculated risk levels ranging between 9×10^{-5} and 2×10^{-3} increased individual lifetime cancer risk. Similarly, nickel and vanadium were predicted to exceed benchmark levels for all three scenarios. Cadmium and chromium exceeded benchmark levels for the commercial landfill scenario only.

The Monte Carlo simulation results strongly corroborated the conservatism of the high-end analyses for oil-fired utility wastes. For all constituents found to exceed the benchmark value in the deterministic analyses, the high-end predicted concentration equaled or exceeded the corresponding

90th percentile concentration predicted in the Monte Carlo simulation,[7] except for vanadium in the onsite monofill scenario (76th percentile). Generally, the high-end exceedence levels corresponded to the 98th percentile or greater Monte Carlo results.

The high-end results suggested higher risk arising from the surface impoundment scenario than for the landfill scenario. Moreover, the risks from the commercial landfill exceeded those for the impoundment for all metals except vanadium. Consideration of the 50th and 95th percentile Monte Carlo results showed the same pattern of expected risk between scenarios.

It is important to note that the total quantity of oil-fired utility wastes contained in the onsite and offsite landfills was the same. The difference in calculated risks resulted directly from the difference in leachate volume generated from each scenario, calculated as surface area times infiltration rate. The quantity of leachate generated in the monofill was 475 cubic meters per year, while the quantity of leachate generated in the codisposal scenario was 3,500 cubic meters. However, each unit was described as having the same starting leachate concentration (i.e. undiluted by the presence of any other materials in the unit). In effect, EPACMTP accelerated the leaching of oil ash wastes by attributing the ash leachate characteristics to the entire leachate flux of the larger commercial landfill, relative to the monofill. This computational characteristic also had the effect of increasing the sensitivity of the model to leachate characteristics. Results for the commercial landfill revealed that the high-end risks exceeded the 99th percentile results from the Monte Carlo analysis for arsenic, barium, cadmium, chromium, copper, nickel, selenium, and vanadium.

Overall, the modeling results suggested that arsenic, vanadium, and potentially nickel remain as constituents of concern for oil-fired utility wastes. Arsenic exceeded the threshold risk value of 1×10^{-6} in all three scenarios. Please see Section 7 for a detailed discussion of uncertainty.

5.4 FLUIDIZED BED COMBUSTION (FBC) WASTES[8]

[7] The Monte Carlo simulation has not been completed for As and Se forthe oil ash surface impoundment.

[8]Beryllium appears throughout this section as a constituent of concern, based on the CSF-based HBL. In April 1998, EPA vacated that CSF. Alternative benchmarks are presented in Appendix O.

5.4.1 Screening Analysis

Table 5-14. Screening Results for FBC Utility Comanaged Wastes

Constituent	HBL[1] (mg/l)	95th % Observed Concentration (mg/l)	Screening Result and Conclusion[2]
Antimony	0.021	1.29	***61.4***
Arsenic	0.00029-(c)	0.35	***1,207***
Barium	3.6	2.6	0.72
Beryllium	0.0001-(c)	0.28	***2,800***
Cadmium	0.026	0.09	***3.46***
Chromium III/VI	0.26	0.29	***1.12***
Copper	1.3-(a)	0.16	0.12
Lead	0.015-(a)	0.49	***32.7***
Mercury	0.015	0.01	0.67
Nickel	1.03	0.41	0.41
Silver	0.257	0.13	0.51
Selenium	0.257	0.26	***1.01***
Thallium	0.0041	0.07	***17.1***
Vanadium	0.36	1.64	***4.56***
Zinc	15.4	4.46	0.29

Notes:
1. All HBLs listed above are non-carcinogens, except for arsenic and beryllium -(c); lead and copper are action levels -(a), not health based numbers.
2. Numbers in **bold** and *italics* indicate which constituents exceed the screening criteria.

Table 5-14 shows screening analysis results for FBC utility wastes. The screening analysis was conducted as described in the preceding sections of this report. The 95th percentile concentrations for barium, copper, mercury, nickel, silver, and zinc were below the screening criteria. Accordingly, these constituents were eliminated from further consideration. All other EPACMTP constituents survived for analysis in modeling.

5.4.2 High-end Analysis

High-end deterministic modeling was conducted for all surviving EPACMTP constituents for FBC wastes using two distinct waste management scenarios: a landfill and a minefill. This modeling

was conducted as described in the preceding sections of this report. The results for each scenario are presented individually below.

Table 5-15. Results for Fluidized Bed Combustion Waste Landfill (Scenario FL), High-end Analysis

Constituent	HBL[1] (mg/l)	95th % Observed Conc. (mg/l)	DAF[2] Result from Modeling	HQ and Conclusion[3,5]
Antimony	0.021	1.29	3.12	***19.69***
Arsenic	0.00029-(c)	0.35	2.15	***Risk = 5.61×10^{-4}***
Beryllium	0.0001-(c)	0.28	2.04	***Risk = 1.37×10^{-3}***
Cadmium	0.026	0.09	17,923	0.00019
Chromium III	0.26	0.29	>10^{6}	<10^{-5}
Chromium VI			1.7	0.66
Lead[4]	0.015	0.49	>10^{42}	<10^{-42}
Selenium	0.257	0.26	1.68	0.60
Thallium	0.0041	0.07	287	0.059
Vanadium	0.36	1.64	482,000	0.0000095

Notes:
1. All HBLs listed above are non-carcinogens, except for arsenic and beryllium -(c).
2. DAF result is the ratio of the initial concentration to the receptor well peak concentration.
3. HQ is predicted receptor well peak concentration divided by HBL.
4. Lead is an action level, not a health based number.
5. Numbers in **bold** and *italics* indicate which constituents exceed the risk threshold (HQ = 1 or risk = 10^{-6}).

Landfill (Scenario FL)

Table 5-15 shows high-end analysis results for the FBC waste landfill (Scenario FL). Antimony, arsenic, and beryllium exceeded the risk threshold under the assumptions used for this scenario. Beryllium exceeded the threshold by a factor of approximately 1,000, arsenic by a factor of over 500, and antimony by a factor of 20. For chromium III and lead, predicted receptor well concentrations were near or equal to zero at all times within the modeling period (10,000 years). All other constituents also were below the risk threshold, with only chromium VI and selenium approaching the threshold (66 and 60 percent of the threshold, respectively). These results are discussed in greater detail in Section 5.4.4.

Minefill (Scenario FF)

Table 5-16. Results for Fluidized Bed Combustion Waste Minefill (Scenario FF), High-end Analysis

Constituent	HBL[1] (mg/l)	95th % Observed Conc. (mg/l)	DAF[2] Result from Modeling	HQ and Conclusion[3,5]
Antimony	0.021	1.29	6.94	***8.85***
Arsenic	0.00029-(c)	0.35	3.58	***Risk = 3.37×10^{-4}***
Beryllium	0.0001-(c)	0.28	2.20	***Risk = 1.27×10^{-3}***
Cadmium	0.026	0.09	27,685	0.00013
Chromium III	0.26	0.29	>10^6	<10^{-5}
Chromium VI			2.42	0.46
Lead[4]	0.015	0.49	>10^6	<10^{-5}
Selenium	0.257	0.26	2.37	0.43
Thallium	0.0041	0.07	373	0.046
Vanadium	0.36	1.64	26,500	0.00017

Notes:
1. All HBLs listed above are non-carcinogens, except for arsenic and beryllium -(c).
2. DAF result is the ratio of the initial concentration to the receptor well peak concentration.
3. HQ is predicted receptor well peak concentration divided by HBL.
4. Lead is an action level, not a health based number.
5. Numbers in **bold** and *italics* indicate which constituents exceed the risk threshold (HQ = 1 or risk = 10^{-6}).

Table 5-16 shows high-end analysis results for the FBC waste minefill (Scenario FF). As they did for the FBC waste landfill, antimony, arsenic, and beryllium exceeded the risk threshold in this scenario. The magnitude of the exceedence for antimony was smaller in this scenario (9 times the threshold), while exceedences for arsenic and beryllium did not change dramatically. For chromium III and lead, predicted receptor well concentrations were near or equal to zero at all times within the modeling period (10,000 years). All other constituents also were below the risk threshold, with only chromium VI and selenium showing an appreciable percentage of the threshold (46 and 43 percent of the threshold, respectively). These results are discussed in greater detail in Section 5.4.4.

5.4.3 Central Tendency (Monte Carlo) Analysis

Central tendency analysis was conducted using probabilistic Monte Carlo modeling for the FBC waste landfill scenario for all surviving EPACMTP constituents. This modeling was conducted as described in the preceding sections of this report.

Landfill (Scenario FL)

Table 5-17. Comparison of Probabilistic and Deterministic Risk Results for FBC Utility Waste Landfill (Scenario FL)

Constituent	Predicted HQ[1] or Risk			
	Monte Carlo		High-end Analysis	
	50th percentile[2]	95th percentile[2]	Result[2]	Corresponding Monte Carlo Percentile
Antimony	0.000000000170	***1.70***	***19.69***	>100th
Arsenic (risk)	1.54×10^{-14}	$\boldsymbol{2.12\times10^{-5}}$	$\boldsymbol{5.61\times10^{-4}}$	99th
Beryllium (risk)	1.28×10^{-15}	$\boldsymbol{1.63\times10^{-5}}$	$\boldsymbol{1.37\times10^{-3}}$	>100th
Cadmium	$<10^{-16}$	0.0321	0.00019	89th
Chromium VI	0.000000000137	0.0457	0.66	100th
Lead	0	0.000000431	$<10^{-42}$	64th
Selenium	0.00000000261	0.0542	0.60	100th
Thallium	0.000000000001	0.498	0.059	91st
Vanadium	0	0.00445	0.00001	92nd

Notes:
1. Values shown are HQs except for arsenic and beryllium, which are risk. HQ is the predicted well concentration divided by the HBL.
2. Numbers in **bold** and *italics* indicate which constituents exceed the risk threshold (HQ=1 or risk=10^{-6}) in a given percentage of model runs.

Table 5-17 presents the probabilistic risk results for the FBC waste landfill (Scenario FL) and compares them to the deterministic results. At the 50th percentile, Monte Carlo modeling predicted all constituents well below the risk threshold. At the 95th percentile, antimony, arsenic, and beryllium exceeded the risk threshold. These 95th percentile exceedences were of lesser magnitude, however, than those found in deterministic modeling (approximately 20 times for arsenic and beryllium and 70 percent for antimony). All other constituents remained below the risk threshold at the 95th percentile of the probabilistic results.

The deterministic high-end results for all constituents except cadmium and lead correspond to the 90th or greater percentile of the Monte Carlo results. Cadmium corresponded to the 89th percentile. For lead, predicted receptor well concentrations were near or equal to zero for nearly all of the Monte Carlo runs (as they were in the deterministic case). Therefore, the percentile rank of

the deterministic results for lead is not meaningful. These results are discussed in greater detail in Section 5.4.4.

5.4.4 Discussion of Results

The screening of FBC wastes indicated generally lower concentrations of constituents of concern compared with comanaged wastes. Modeling results showed that very few constituents of concern were predicted to exceed benchmark concentrations in near-by down-gradient ground water. Arsenic was predicted to exceed benchmark levels in both FBC management scenarios, with calculated risk levels ranging between 3×10^{-4} and 6×10^{-4} increased individual lifetime cancer risk. Similarly, beryllium was predicted to result in an increased cancer risk ranging between 1.3×10^{-3} and 1.4×10^{-3}. Antimony was predicted to exceed its benchmark in both scenarios as well. Predicted risks for the minefill scenario were less than those for the landfill, consistent with the smaller fill area and total capacity, and correspondingly total lower flux to the subsurface.

The Monte Carlo simulation (performed for the landfill only) results strongly corroborated the conservatism of the high-end analyses for FBC wastes. For all three constituents found to exceed the benchmark value in the landfill deterministic analyses, the high-end predicted concentration equaled or exceeded the corresponding 99th percentile concentration predicted in the Monte Carlo simulation. This finding was consistent with other high-end landfill scenarios that generally show the selection of well location and constituent concentration to yield a high estimate of risk compared with the 95th percentile result for the Monte Carlo simulation for most metals.

Overall, the modeling results suggested that arsenic and beryllium remain as constituents of concern. The predicted high-end hazard quotient for antimony from the FBC landfill scenario was roughly 20. This compares with roughly 9 for the minefill scenario. Further, the 95th percentile result from the Monte Carlo simulation for the landfill predicted an HQ of less than 2. Although these results suggest modest risk, antimony also was retained for further consideration as a constituent of concern. Please see Section 7 for a detailed discussion of uncertainty.

5.5 NON-UTILITY COMBUSTION WASTES

5.5.1 Screening Analysis

Table 5-18. Results for Non-utility Coal Combustion Waste Onsite Monofill (Scenario NM), High-end Analysis

Constituent	HBL[1] (mg/l)	95th % Observed Conc. (mg/l)	DAF[2] Result from Modeling	HQ and Conclusion[3,5]
Arsenic	0.00029-(c)	9.64	4.61	***Risk = 7.21×10^{-3}***
Barium	3.60	27.4	4.18	***1.82***
Cadmium	0.026	0.156	6.10	0.98
Chromium III	0.26	0.746	>10^{6}	<10^{-5}
Chromium VI			4.21	0.68
Lead[4]	0.015	0.468	>10^{18}	<10^{-18}
Nickel	1.03	8.33	4.18	***1.93***
Selenium	0.257	1.03	6.02	0.66
Vanadium	0.360	0.800	51,000	0.000044
Zinc	15.4	23.1	8.37	0.18

Notes:
1. All HBLs listed above are non-carcinogens, except for arsenic -(c).
2. DAF result is the ratio of the initial concentration to the receptor well peak concentration.
3. HQ is predicted receptor well peak concentration divided by HBL.
4. Lead is an action level, not a health based number.
5. Numbers in **bold** and *italics* indicate which constituents exceed the risk threshold (HQ =1).

Utility comanaged waste characterization data were used for non-utilities, because of a lack of available data for non-utility waste. Therefore, a separate screening analysis was not conducted. All EPACMTP constituents that survived the screening analysis for coal-fired utility comanaged waste (see Section 5.2.1) were analyzed in modeling for non-utilities.

5.5.2 High-end Analysis

High-end deterministic modeling was conducted for all surviving EPACMTP constituents for non-utility combustion wastes using two distinct waste management scenarios: a monofill and a commercial landfill. This modeling was conducted as described in the preceding sections of this report. The results for each scenario are presented individually below.

Onsite Monofill (Scenario NM)

Table 5-19. Results for Non-utility Coal Combustion Waste Commercial Landfill (Scenario NL), High-end Scenario

Constituent	HBL[1] (mg/l)	95th % Observed Conc. (mg/l)	DAF[2] Result from Modeling	HQ and Conclusion[3,5]
Arsenic	0.00029-(c)	9.64	2.80	***Risk = 1.19×10^{-2}***
Barium	3.60	27.4	2.20	***3.46***
Cadmium	0.026	0.156	4.20	***1.43***
Chromium III	0.26	0.746	>10^{6}	<10^{-5}
Chromium VI			2.20	***1.30***
Lead[4]	0.015	0.468	>10^{18}	<10^{-18}
Nickel	1.03	8.33	2.20	***3.68***
Selenium	0.257	1.03	4.86	0.82
Vanadium	0.360	0.800	16,000	0.00014
Zinc	15.4	23.1	4.38	0.34

Notes:
1. All HBLs listed above are non-carcinogens, except for arsenic -(c).
2. DAF result is the ratio of the initial concentration to the receptor well peak concentration.
3. HQ is predicted receptor well peak concentration divided by HBL.
4. Lead is an action level, not a health based number.
5. Numbers in **bold** and *italics* indicate which constituents exceed the risk threshold (HQ =1).

Table 5-18 shows high-end analysis results for the non-utility combustion waste monofill (Scenario NM). Three constituents exceeded the risk threshold under the assumptions used for this scenario: arsenic, barium, and selenium. Arsenic exceeded the threshold by approximately 7,000 times. Barium and selenium were each just under twice the threshold. Cadmium, chromium VI, and selenium all approached the threshold, at 98, 68, and 66 percent of their respective HBLs. Predicted receptor well concentrations for chromium III and lead were near or equal to zero at all times within the modeling period (10,000 years). These results are discussed in greater detail in Section 5.5.4.

Offsite Commercial Landfill (Scenario NL)

Table 5-19 shows high-end analysis results for the non-utility combustion waste commercial landfill (Scenario NL). Several constituents exceeded the risk threshold under the assumptions used for this scenario: arsenic, barium, cadmium, chromium VI, and nickel. Arsenic exceeded the threshold by about four orders of magnitude. Barium and nickel were approximately three times the threshold, while exceedences for cadmium and chromium VI were 43 and 30 percent, respectively.

Predicted receptor well concentrations for chromium III and lead were near or equal to zero at all times within the modeling period (10,000 years). Of the other constituents that did not exceed the threshold, selenium was closest to its HBL (82 percent). These results are discussed in greater detail in Section 5.5.4.

5.5.3 Central Tendency (Monte Carlo) Analysis

Central tendency analysis was conducted using probabilistic Monte Carlo modeling for both of the non-utility combustion waste scenarios (monofill and commercial landfill) for all surviving EPACMTP constituents. This modeling was conducted as described in the preceding sections of this report. The results for each of these two scenarios are presented individually below.

Table 5-20. Comparison of Probabilistic and Deterministic Risk Results for Non-utility Combustion Waste Monofill (Scenario NM)

	Predicted HQ[1] or Risk			
	Monte Carlo		High-end Analysis	
Constituent	50th percentile[2]	95th percentile[2]	Result[2]	Corresponding Monte Carlo Percentile
Arsenic (risk)	5.81×10^{-7}	***2.60×10^{-4}***	***Risk = 7.21×10^{-3}***	99th
Barium	0.00017	0.020	***1.82***	99th
Cadmium	0.0011	0.14	0.98	99th
Chromium	0.00044	0.099	0.68	99th
Lead	0.00000	0.0028	$<10^{-18}$	82nd
Nickel	0.000001	0.027	***1.93***	>100th
Selenium	0.0013	0.12	0.66	99th
Vanadium	0.0000	0.14	0.000044	90th
Zinc	$<10^{-13}$	0.00097	0.18	99th

Notes:
1. Values shown are HQs except for arsenic and beryllium, which are risk. HQ is the predicted well concentration divided by the HBL.
2. Numbers in **bold** and *italics* indicate which constituents exceed the risk threshold (HQ=1 or risk=10^{-6}) in a given percentage of model runs.

Onsite Monofill (Scenario NM)

Table 5-20 presents the probabilistic risk results for the non-utility combustion waste monofill (Scenario NM) and compares them to the deterministic results. At the 50th percentile, Monte Carlo

modeling predicted all constituents below the risk threshold. At the 95th percentile, only arsenic exceeded the risk threshold. All other constituents remained below the risk threshold at the 95th percentile of the probabilistic results.

The deterministic high-end results for all constituents except lead correspond to the 90th or greater percentile of the Monte Carlo results. In fact, nearly half of the deterministic results correspond to the 99th or greater percentile of the Monte Carlo runs. For lead, predicted receptor well concentrations were near or equal to zero for nearly all of the Monte Carlo runs (as they were in the deterministic case). Therefore, the percentile rank of the deterministic results for lead is not meaningful. These results are discussed in greater detail in Section 5.5.4.

Offsite Commercial Landfill (Scenario NL)

Table 5-21 presents the probabilistic risk results for the non-utility combustion waste commercial landfill (Scenario NL) and compares them to the deterministic results. At the 50th percentile, Monte Carlo modeling predicted all constituents below the risk threshold. At the 95th percentile, only arsenic exceeded the risk threshold. All other constituents remained below the risk threshold at the 95th percentile of the probabilistic results.

Table 5-21. Comparison of Probabilistic and Deterministic Risk Results for Non-utility Combustion Waste Commercial Landfill (Scenario NL)

Constituent	Predicted HQ or Risk			
	Monte Carlo		High-end Analysis	
	50th percentile	95th percentile	Result	Corresponding Monte Carlo Percentile
Arsenic (risk)	9.18×10^{-7}	***3.74×10^{-4}***	***1.19×10^{-2}***	>100th
Barium	0.00029	0.024	***3.46***	>100th
Cadmium	0.00000014	0.028	***1.43***	99th
Chromium	0.00084	0.13	***1.30***	>100th
Lead	0.0000	0.00018	$<10^{-18}$	82nd
Nickel	0.000014	0.040	***3.68***	>100th
Selenium	0.00091	0.067	0.82	99th
Vanadium	0.0000000	0.022	0.00014	86th
Zinc	0.000000000	0.0012	0.34	99th

Notes:
1. Values shown are HQs except for arsenic and beryllium, which are risk.
2. Numbers in **bold** and *italics* indicate which constituents exceed the risk threshold (HQ=1 or risk=10^{-6}).

The deterministic high-end results for all constituents except lead and vanadium correspond to the 90th or greater percentile of the Monte Carlo results. In fact, nearly half of the deterministic results exceed the 100th percentile of the Monte Carlo runs. Vanadium's results correspond to the 86th percentile. For lead, predicted receptor well concentrations were near or equal to zero for nearly all of the Monte Carlo runs (as they were in the deterministic case). Therefore, the percentile rank of the deterministic results for lead is not meaningful. These results are discussed in greater detail in Section 5.5.4.

5.5.4 Discussion of Results

As discussed above, EPA used the characterization data for utility comanaged wastes to describe non-utility wastes for modeling purposes. As with the comanagement scenarios, modeling results showed that very few constituents of concern were predicted to exceed benchmark concentrations in near-by down-gradient ground water. Arsenic was predicted to exceed benchmark levels in both on-site and commercial non-utility landfill scenarios, with calculated risk levels ranging

between 7×10^{-3} and 1.2×10^{-2} increased individual lifetime cancer risk. Several other metals (barium, cadmium, chromium, and nickel) were predicted to exceed their respective benchmark by a factors of 3.7 or less in one or both management scenarios. Predicted risks for the commercial landfill exceeded those for the monofill, consistent with the oil-fired utility waste scenarios. Predicted risks were generally lower than those predicted for the comanaged waste landfill, consistent with the significantly smaller size of the non-utility landfills.[9]

The Monte Carlo simulation results strongly corroborated the conservatism of the high-end analyses for non-utility waste management scenarios. For all constituents found to exceed the benchmark value in the monofill and commercial landfill deterministic analyses, the high-end predicted concentration equaled or exceeded the corresponding 99th percentile concentration predicted in the Monte Carlo simulation. This finding was consistent with other high-end landfill scenarios that generally show the selection of well location and constituent concentration to yield a high estimate of risk compared with the 95th percentile result for the Monte Carlo simulation for most metals. In fact, at the 95th percentile level, only arsenic was found to exceed the risk threshold in either scenario.

Overall, the modeling results suggested that arsenic remains as a constituent of concern. The predicted high-end hazard quotient for all other metals was less than 4 in both scenarios. Further, cadmium and chromium only exceeded threshold values in the commercial landfill scenario, which magnifies the leachate contribution to the subsurface.[10] Given the modest risk suggested by these results, barium, cadmium, chromium, and nickel were dropped from further consideration for these wastes. Please see Section 7 for a detailed discussion of uncertainty.

5.6 TIME TO REACH CONCENTRATIONS OF CONCERN

[9] The exception was cadmium, which was found to exhibit significantly higher mobility in this scenario than in the coal-fired utility co-managed waste scenario. This is believed to result computational errors associated with the non-linear adsorption isotherm for cadmium, and could not be resolved before the release of this draft document.

[10] See Section 7 for more detailed discussion of the difficulties with the commercial scenarios as modeled.

Table 5-22 shows all constituents that exceed their respective risk thresholds in each scenario. The results demonstrate that certain constituents require additional consideration. As can be seen, arsenic was found to exceed the HBL in near-by ground water for all scenarios. Nickel and vanadium exceeded their HBLs in all oil ash scenarios and several other metals occasionally were predicted to exceed threshold risk levels in landfill and impoundment scenarios.

For all modeled scenarios, the receptor location was set at 150 meters directly downgradient from the unit boundary. In addition, for all scenarios, the study period was set at 10,000 years. Finally, for all scenarios, the risk was defined to be the ratio of the peak concentration to the HBL observed at any time during the study period. Conceptually, however, risks predicted based on a peak concentration in a downgradient well within a few years are suject to considerably lower uncertainty than risks predicted on a concentration predicted to occur in thousands of years, all else being equal. Accordingly, this section presents the results of an analysis of the time to reach the threshold risk (C_t = HBL) in downgradient wells for all constituents predicted to exceed the threshold risk at some time during the study period in the high-end deterministic analyses.

The results show that every constituent in the landfill and minefill scenarios generally requires a long period of time (greater than 1,000 years) to reach concentration of concern at plume centerline at a receptor well distance of 150 meters, with the exception of vanadium. In contrast, constituents of concern in coal comanagement and oil combustion waste impoundments reach a receptor in 500 years or less, and of particular note is vanadium, which reaches a downgradient receptor well within the operating period of the management unit.

Table 5-22. Hazard Quotients for High-End Analyses: Constituents Predicted to Exceed HBLs in Downgradient Ground Water Under Assumptions Modeled

Constituent	Coal-fired utility comanagement			Oil-fired utility			FBC		Nonutiity	
	Surface Imp.	Landfill	Minefill	Surface Imp.	On-site monofill	Off-site landfill	Landfill	Minefill	On-site monofill	Off-site landfill
Antimony	N/A[3]			N/A[1]			19.69	8.85	N/A[3]	
Arsenic (risk)	$5.08x10^{-4}$	$1.05x10^{-2}$	$8.52x10^{-3}$	$2.56x10^{-4}$	$9.1x10^{-5}$	$2.37x10^{-3}$	$5.61x10^{-4}$	$3.37x10^{-4}$	$7.21x10^{-3}$	$1.19x10^{-2}$
Barium	N/A[2]					1.06	N/A[2]		N/A[2]	N/A[2]
Beryllium	N/A[1]			N/A[1]			$1.37x10^{-3}$	$1.27x10^{-3}$	N/A[1]	
Cadmium						5.24				1.43
Chromium		1.56	1.28			2.46				1.30
Copper	N/A[2]						N/A[2]		N/A[2]	
Lead										
Mercury	N/A[2]						N/A[2]		N/A[2]	
Nickel		4.57	3.53	101	3.43	135	N/A (2)		1.93	3.68
Selenium		1.78	1.42							
Silver	N/A[3]			N/A[2]			N/A[2]		N/A[3]	
Thallium	N/A[1]			N/A[1]					N/A[1]	
Vanadium				946	8.33	364				
Zinc				N/A[2]			N/A[2]			

Notes:
1. Insufficient data to support assessment.
2. Did not exceed screening levels and thus was not carried forward to high-end determiniatic and probabilistic modeling (others).
3. All samples below detection limits, so not carried forward to high-end deterministic and probabilistic modeling.

No entry: Results were below threshold risk levels (HQ< 1).

Table 5-23. Comparison of the Predicted Time to Reach Risk for All High-end Deterministic Scenarios (years)

Constituent	HBL (mg/l)	Years to Reach Health Based Level for Scenario:									
		Coal-firedutility comanagement			Oil-fired utility			FBC		Nonutility	
		Surface Imp. CS	Landfill CL	Minefill CF	Surface Imp. OS	Onsite monofill OM	Offsite Landfill OL	Landfill FL	Minefill FM	Onsite Monofill NM	Offisite Landfill NL
Antimony	0.021							5,800	6,700		
Arsenic	0.00029-(c)	500	2,800	2,900	400	2,800	2,000	3,600	3,700	1,400	1,300
Barium	3.6						2,800				
Beryllium	0.0001-(c)							6,500	1,400		
Cadmium	0.026						2,500				8,900
Chromium VI	0.26		4,700	7,000			3,300				3,700
Copper											
Lead											
Mercury											
Nickel	1.03		6,200	2,900	50	900	200			1,500	1,400
Selenium	0.257		1,500	1,800							
Silver											
Thallium											
Vanadium	0.36				10	80	70				
Zinc											

Notes:
1. Risk defined as C_t = HBL
2. Blank speces indicate that no risk predicted

6. RISKS TO CHILDREN

6.1 OVERVIEW

This section evaluates the potential risks for child receptors from exposure to ground-water contamination resulting from remaining FFC waste management. This section focuses on children because they are expected to be a more sensitive subpopulation. Note that EPA has not yet developed a policy detailing the manner in which child populations should be considered in national risk assessments. This assessment considers two specific child populations: a young group, ranging in age from 1 to 10 years, and a larger group, ranging in age from 1 to 19 years. The method followed here was intended to identify the potential difference in risk associated with these populations, based solely on different exposure profiles. No attempt was made to adjust toxicity values to make them child-specific. In fact, the toxicity values (RfD and CSF) used in this assessment may already reflect lifetime cancer risks and/or sensitive populations.

To assess potential risks to children, EPA calculated child-specific benchmark values for each constituent using child-specific exposure assumptions different from those for adults discussed in the preceding sections. Section 6.2 describes these calculations. EPA then applied the child-specific benchmarks to the high-end deterministic modeling results presented in Section 5 to identify changes in the set of constituents exceeding the risk threshold for each modeled scenario. The resulting changes are discussed in Section 6.3.

6.2 CALCULATION OF CHILD-SPECIFIC BENCHMARKS

Child receptors are assumed to be exposed by ingestion of ground water. Exposure resulting from other uses of ground water (i.e., showering) is not considered because showering results in exposure from inhalation, which is not applicable for metals. EPA calculated child-specific benchmarks for each of two child populations (ages 1-10, and ages 1-19). This recalculation was applied only to HBLs. No effort was made to adjust the action level-based benchmarks for lead and copper, because action levels have different toxicity and exposure assumptions than those used for the HBLs in this study.

Benchmarks were recalculated by changing the body weight, tap water intake, and exposure duration assumptions to better reflect child consumers of affected ground water. Specifically, EPA calculated age- and body weight-adjusted average values for average daily tap water intake, age-adjusted average body weights, and age-adjusted exposure durations using the 1996 draft exposure factors handbook (EPA, 1996). Principal assumptions regarding the receptors are presented in Table 6-1. Note that central tendency assumptions are used throughout. Using these assumptions, Table 6-2 below compares the resulting benchmark values for children from ages 1 to 10 and for children from ages 1 to 19 to those for adults used in Section 5. The calculations used to derive these benchmarks are discussed in Appendix B.

Table 6-1. Assumptions for Adult and Child Residents (Both Sexes) Scenario, Ground-Water Ingestion Pathway

Exposure Parameter	Value Used			Source
	Adult	Children (age 1–10)	Children (age 1–19)	
Ingestion rate (L/d)	1.4	0.74	0.82	Mean ingestion rate[1]
Exposure duration (yr)	9	6.5	7.2	Median residence time[2]
Body weight (kg)	72	22.0	38.3	Mean body weight[3]
Lifetime (yr)	75	75	75	Mean life expectancy[4]
Exposure frequency (days/yr)	350	350	350	Assumed

Source: *EPA Draft Exposure Factors Handbook*, 1996.
Notes:
1. Derived from Tables 3-7 and 3-10.
2. Derived from Table 14-159 and page 14-16.
3. From Tables 7-2 and 7-10.
4. From page 8-1.

As shown in Table 6-2, all of the benchmark values for children from ages 1 to 10 are less than the corresponding values for adults. Thus, this population is more sensitive than adults to all the constituents evaluated in this report because of their lower mean body weight. The benchmark values for children from ages 1 to 19 fall between those for adults and those for children ages 1 to 10, with two exceptions: arsenic and beryllium. The benchmarks for children from ages 1 to 19 for these two carcinogens are greater than those for adults. Therefore, this older age group is slightly less susceptible to these two constituents, because lesser exposure duration and ingestion rate outweighs decreased body weight for this group. For all other constituents, the age 1 to 19 benchmarks are

slightly less than the corresponding adult benchmarks. Like the youngest group, therefore, this group also is more sensitive to most constituents.

Table 6-2. Comparison of Adult and Child Health-based Benchmark Values Derived for the Remaining FFC Waste Risk Assessment

Constituent	RfD[1] (mg/kg/day)	Carcinogen Slope Factor[1] $(mg/kg/day)^{-1}$	Health-based Level[2] (mg/l)		
			Adult	Children (age 1–10)	Children (age 1–19)
Antimony	0.0004	–	0.021	0.012	0.019
Arsenic	0.0003	1.5	0.00029-(c)	0.00023-(c)	0.00033-(c)
Barium	0.07	–	3.60	2.088	3.24
Beryllium	0.005	4.3	0.0001-(c)	0.00008-(c)	0.00011-(c)
Cadmium	0.0005	–	0.026	0.015	0.0234
Chromium VI	0.005	–	0.26	0.151	0.234
Mercury	0.0003	–	0.015	0.0087	0.0135
Nickel	0.02	–	1.03	0.597	0.927
Selenium	0.005	–	0.257	0.149	0.231
Silver	0.005	–	0.257	0.149	0.231
Thallium	0.00008	–	0.0041	0.0024	0.0037
Vanadium[3]	0.007	–	0.360	0.209	0.324
Zinc	0.3	–	15.4	8.93	13.9

Notes:
1. Source of RfDs and CSFs is IRIS (June, 1997) unless otherwise noted.
2. All HBLs listed above are non-carcinogens, except for arsenic and beryllium -(c).
3. RfD for Vanadium is from HEAST.

6.3 APPLICATION OF CHILD-SPECIFIC BENCHMARKS AND DISCUSSION OF RESULTS

The generally lower benchmarks for children confirm that they are a more sensitive subpopulation, as expected. For each of the high-end deterministic scenarios modeled in Section 5, Tables 6-3 through 6-12 show the changing outcomes when the child-specific benchmarks were applied.

Children from ages 1 to 19 appear slightly less sensitive to arsenic and beryllium than adults. Therefore, the risks from these two constituents decreased slightly for this age group. This decrease, however, did not result in a meaningful difference for these two constituents. When arsenic or beryllium exceeded the risk threshold for adults, they did so by a large margin. Thus, the slight

decrease in risk did not change the set of constituents exceeding the risk threshold for children from ages 1 to 19.

All of the other HQs and risks increased when evaluated with respect to children, because of the greater sensitivity described above. With a few exceptions, however, these increases did not change the set of constituents exceeding the risk threshold. In a few cases, these increases did result in additional constituents above the threshold for children from ages 1 to 10. These cases were those where the adult HQs were within 60 percent of the risk threshold. In one case, where the adult HQ was very close to the risk threshold, the increases also resulted in an additional constituent exceeding the threshold for children from ages 1 to 19. The specific scenarios with additional exceedences for children are summarized below:

- Zinc in the coal-fired utility minefill (Scenario CF),
- Chromium VI and selenium in the FBC waste landfill (Scenario FL),
- Cadmium, chromium VI, and selenium in the non-utility combustion waste monofill (Scenario NM), and
- Selenium in the non-utility combustion waste commercial landfill (Scenario NL).

With the exception of cadmium in the non-utility monofill, the above constituents are of concern for the youngest group of children (ages 1 to 10) only. This small group of constituents is not a substantial addition to those identified in Section 5. The exceedence for zinc is believed to be the result of model instability around the input concentration used. In going from the coal-fired utility comanaged waste landfill to the minefill, every other constituent showed a decrease in mobility, which was expected given the latter scenario's lesser surface area. Zinc's mobility, however, increased. This counter-intuitive result is believed to be caused by computational errors associated with the non-linear isotherm for zinc.

For the other constituents, the additional exceedences are, in all cases, small (less than twice the threshold). Furthermore, the Monte Carlo modeling results for these scenarios (see Section 5) show the high-end values for cadmium, chromium VI, and selenium to be conservative (in the 99th or greater percentile). The 95th percentile Monte Carlo results for these constituents in these scenarios fall far below the risk threshold, even when compared to the more sensitive child-specific

benchmarks.[1] Note also that there are significant uncertainties surrounding cadmium and chromium VI in particular (see Section 7).

Therefore, this analysis demonstrates slightly higher risks to a sensitive subpopulation. Consideration of this subpopulation, however, does not result in the identification of any additional constituents of concern beyond those previously identified in Section 5.

Table 6-3. Results for a Coal-fired Utility Comanaged Waste Surface Impoundment (Scenario CS)

Constituent	High-end HQ[1]		
	Adult Resident	Child Resident (age 1–10)	Child Resident (age 1–19)
Antimony	–	–	–
Arsenic (risk)	***5.08×10^{-4}***	***6.41×10^{-4}***	***4.47×10^{-4}***
Barium	0.242	0.418	0.269
Cadmium	0.028	0.048	0.031
Chromium VI	0.069	0.120	0.077
Mercury	–	–	–
Nickel	0.085	0.146	0.094
Selenium	0.368	0.634	0.409
Silver	–	–	–
Thallium	–	–	–
Vanadium	<1.0×10^{-19}	<3.83×10^{-19}	<2.47×10^{-19}
Zinc	0.013	0.022	0.014

Note:
1. Values shown are HQ's, except for arsenic, which is risk. Numbers in bold and italics indicate which constituents exceed the risk threshold (HQ=1 or risk=10^{-6}).

[1] The relevant 95th percentile HQs, adjusted to child-specific benchmarks, are:
Scenario FL -- chromium VI, 0.078; selenium, 0.094
Scenario NM -- cadmium, 0.24; chromium VI, 0.17; selenium, 0.21
Scenario NL -- selenium, 0.12

Table 6-4. Results for a Coal-fired Utility Comanaged Waste Landfill (Scenario CL)

Constituent	High-end HQ[1]		
	Adult Resident	Child Resident (age 1–10)	Child Resident (age 1–19)
Antimony	–	–	–
Arsenic (risk)	***1.05×10^{-2}***	***1.32×10^{-2}***	***9.22×10^{-3}***
Barium	***4.32***	***7.46***	***4.81***
Cadmium	0.00025	0.00044	0.00028
Chromium VI	***1.56***	***2.69***	***1.73***
Mercury	--	--	--
Nickel	***4.57***	***7.88***	***5.08***
Selenium	***1.78***	***3.07***	***1.98***
Silver	--	--	--
Thallium	--	--	--
Vanadium	1.2×10^{-12}	3.83×10^{-12}	2.47×10^{-12}
Zinc	0.000000088	0.000000051	0.000000079

Note:
1. Values shown are HQ's, except for arsenic, which is risk. Numbers in bold and italics indicate which constituents exceed the risk threshold (HQ=1 or risk=10^{-6}).

Table 6-5. Results for a Coal-fired Utility Comanaged Waste Minefill (Scenario CF)

Constituent	High-end HQ[1]		
	Adult Resident	Child Resident (age 1–10)	Child Resident (age 1–19)
Antimony	–	–	–
Arsenic (risk)	***8.52×10^{-3}***	***1.07×10^{-2}***	***7.49×10^{-3}***
Barium	***3.51***	***6.05***	***3.90***
Cadmium	0.00023	0.00039	0.00025
Chromium VI	***1.28***	***2.20***	***1.42***
Mercury	–	–	–
Nickel	***3.53***	***6.09***	***3.92***
Selenium	***1.42***	***2.44***	***1.58***
Silver	–	–	–
Thallium	–	–	–
Vanadium	$<1.0\times10^{-10}$	$<3.83\times10^{-9}$	$<2.47\times10^{-9}$
Zinc	0.69	***1.19***	0.76

Note:
1. Values shown are HQ's, except for arsenic, which is risk. Numbers in bold and italics indicate which constituents exceed the risk threshold (HQ=1 or risk=10^{-6}).

Table 6-6. Results for an Oil-fired Utility Waste Surface Impoundment (Scenario OS)

Constituent	High-end HQ[1]		
	Adult Resident	Child Resident (age 1–10)	Child Resident (age 1–19)
Antimony	–	–	–
Arsenic (risk)	***2.56×10^{-4}***	***3.22×10^{-4}***	***2.25×10^{-4}***
Barium	0.17	0.30	0.19
Cadmium	0.27	0.46	0.30
Chromium VI	0.37	0.65	0.42
Mercury	0.24	0.42	0.27
Nickel	***101***	***175***	***112***
Selenium	0.15	0.27	0.17
Silver	–	–	–
Thallium	–	–	–
Vanadium	***946***	***1,629***	***1,051***
Zinc	–	–	–

Notes:
1. Values shown are HQ's, except for arsenic, which is risk. Numbers in bold and italics indicate which constituents exceed the risk threshold (HQ=1 or risk=10^{-6}).

Table 6-7. Results for an Oil-fired Utility Waste Onsite Monofill (Scenario OM)

Constituent	High-end HQ[1]		
	Adult Resident	Child Resident (age 1–10)	Child Resident (age 1–19)
Antimony	–	–	–
Arsenic (risk)	***9.06×10^{-5}***	***1.14×10^{-4}***	***7.959×10^{-5}***
Barium	0.024	0.042	0.027
Cadmium	0.11	0.18	0.12
Chromium VI	0.085	0.147	0.095
Mercury	–	–	–
Nickel	***3.43***	***5.92***	***3.81***
Selenium	0.010	0.017	0.011
Silver	–	–	–
Thallium	–	–	–
Vanadium	***8.33***	***14.4***	***9.26***
Zinc	–	–	

Note:
1. Values shown are HQ's, except for arsenic, which is risk. Numbers in bold and italics indicate which constituents exceed the risk threshold (HQ=1 or risk=10^{-6}).

Table 6-8. Results for an Oil-fired Utility Waste Commercial Landfill (Scenario OL)

Constituent	High-end HQ[1]		
	Adult Resident	Child Resident (age 1–10)	Child Resident (age 1–19)
Antimony	–	–	–
Arsenic	***2.37×10^{-3}***	***2.99×10^{-3}***	***2.08×10^{-3}***
Barium	***1.06***	***1.83***	***1.18***
Cadmium	***5.07***	***8.79***	***5.64***
Chromium VI	***2.46***	***4.23***	***2.73***
Mercury	0.00076	0.00131	0.00084
Nickel	***135***	***234***	***150***
Selenium	0.40	0.69	0.45
Silver	–	–	–
Thallium	–	–	–
Vanadium	***364***	***626***	***404***
Zinc	–	–	–

Note:
1. Values shown are HQ's, except for arsenic, which is risk. Numbers in bold and italics indicate which constituents exceed the risk threshold (HQ=1 or risk=10^{-6}).

Table 6-9. Results for a Fluidized Combustion Waste Landfill (Scenario FL)

Constituent	High-end HQ[1]		
	Adult Resident	Child Resident (age 1–10)	Child Resident (age 1–19)
Antimony	***19.7***	***34.5***	***21.8***
Arsenic (risk)	***5.61×10^{-4}***	***7.08×10^{-4}***	***4.93×10^{-4}***
Barium	–	–	–
Beryllium (risk)	***1.37×10^{-3}***	***1.72×10^{-3}***	***1.25×10^{-3}***
Cadmium	0.00019	0.00033	0.00021
Chromium VI	0.66	***1.13***	0.73
Mercury	–	–	–
Nickel	–	–	–
Selenium	0.60	***1.04***	0.67
Silver	–	–	–
Thallium	0.059	0.102	0.066
Vanadium	0.0000095	0.0000163	0.0000105
Zinc	–	–	–

Note:
1. Values shown are HQ's, except for arsenic and beryllium, which are risk. Numbers in bold and italics indicate which constituents exceed the risk threshold (HQ=1 or risk=10^{-6}).

Table 6-10. Results for a Fluidized Combustion Waste Minefill (Scenario FF)

Constituent	High-end HQ[1]		
	Adult Resident	Child Resident (age 1–10)	Child Resident (age 1–19)
Antimony	***8.85***	***15.5***	***9.78***
Arsenic (risk)	***3.37×10^{-4}***	***4.25×10^{-4}***	***2.96×10^{-4}***
Barium	–	–	–
Beryllium (risk)	***1.27×10^{-3}***	***1.59×10^{-3}***	***1.16×10^{-3}***
Cadmium	0.00013	0.00022	0.00014
Chromium VI	0.46	0.79	0.51
Mercury	–	–	–
Nickel	–	–	–
Selenium	0.43	0.74	0.47
Silver	–	–	–
Thallium	0.046	0.078	0.051
Vanadium	0.00017	0.00030	0.00019
Zinc	–	–	–

Note:
1. Values shown are HQ's, except for arsenic and beryllium, which are risk. Numbers in bold and italics indicate which constituents exceed the risk threshold (HQ=1 or risk=10^{-6}).

Table 6-12. Results for Non-utility Combustion Waste Commercial Landfill (Scenario NL)

Constituent	High-end HQ[1]		
	Adult Resident	Child Resident (age 1–10)	Child Resident (age 1–19)
Antimony	–	–	–
Arsenic (risk)	***1.19×10^{-2}***	***1.50×10^{-2}***	***1.04×10^{-2}***
Barium	***3.46***	***5.96***	***3.84***
Cadmium	***1.43***	***2.48***	***1.59***
Chromium VI	***1.30***	***2.25***	***1.45***
Mercury	–	–	–
Nickel	***3.68***	***6.34***	***4.08***
Selenium	0.82	***1.42***	0.92
Silver	–	–	–
Thallium	–	–	–
Vanadium	0.00014	0.00024	0.00015
Zinc	0.34	0.59	0.38

Note:
1. Values shown are HQ's, except for arsenic, which is risk. Numbers in bold and italics indicate which constituents exceed the risk threshold (HQ=1 or risk=10^{-6}).

7. UNCERTAINTY

As with any risk assessment, EPA's study of the risks associated with remaining FFC wastes represents a simplification of reality based on an amalgamation of data and assumptions guided by scientific principles and best professional judgement, and the goal of applying results to a nationwide assessment. Previous sections of the report presented the principal data and assumptions employed in developing the modeling results. This section discusses the primary sources of uncertainty within the risk assessment and the effects that uncertainty has on assessment outcomes. The discussion is intended to provide a basis for developing conclusions regarding the results of modeling. In general, the principal sources of uncertainty fall into four categories:

- Simplification of actual waste composition
- Assumptions affecting leaching rate and volume
- Simplification of fate and transport processes modeled
- Toxicity of contaminants.

Individual issues under each of these broad categories are discussed below.

7.1 SIMPLIFICATION OF ACTUAL WASTE COMPOSITION

The composition of the leachate generated by the wastes assessed in this report is highly variable, and this introduces uncertainty because data from the sampled population may or may not be representative of the total population. The following specific issues are discussed here:

- The representativeness of waste characterization data, specifically the values used as initial leachate concentrations.

- The appropriateness of the analysis methods to simulate leachate composition
- Inherent differences in the characteristics of leachate from impoundments and landfills

Issue: Waste Characterization Data are Limited Relative to the Generator Population Size

The small sample size for waste characteristics data, the high demonstrated variability of waste characteristics, and the demonstrated sensitivity of the model to this parameter present significant uncertainty issues for EPA. Chief among these is the representativeness of the data. Representativeness refers to the confidence that the sample population covers the range and shape of the actual population, and this plays an important role in the applicability of modeling results to the larger population. EPA is not confident that the waste characterization data to represent the range of variability likely to be present in the actual population. Generally, the number of samples evaluated relative to the total number of facilities generating the wastes was highest for the FBC wastes and the oil combustion wastes, and lowest for the comanaged wastes. The comanaged waste characterization data are discussed in more detail below. While the EPRI site investigations included waste characterization data from only 14 comanaged waste sites, the samples (1) reflect conditions at sites burning a wide range of coal types, (2) included the two major utility boiler types of pulverized and cyclone boiler, (3) were collected from locations throughout the country, and (4) covered a wide range of low-volume wastes managed in conjunction with some or all of the large-volume wastes. Further, the sampling plan for many of the sites supports the conclusion that, for some sites, worst case conditions predominated in the sample sets. For example, the EPRI comanagement site investigation sampling plans explicitly directed the identification and sampling of areas within comanagement units most affected by low-volume wastes. For one site, active pyrite oxidation resulting in the formation of strongly acidic leaching conditions controlled the constituent concentrations observed in many samples. As a result, these samples dominated the calculated average for the samples at the site, and the site represented the 95th percentile concentration for all comanaged waste samples for 6 constituents of concern (Appendix F shows the site associated with each 95th percentile concentration used in the risk assessment).

EPA also found specific difficulties with beryllium because it was not sampled at all sites studied. For example, EPA received 11 porewater samples from two sites for which beryllium was analyzed in utility comanaged wastes. Ten samples from one site were all below the method detection limit. The sole sample from the second site showed beryllium to be present at a concentration one and a half times the MCL. EPA could not conclude on the basis of this single sample where beryllium was detected that beryllium variability in comanaged wastes was

represented.

Issue: Porewater Data Were Used to Represent Comanaged Waste Impoundment Leachate

The use of porewater samples to represent leachate also introduces some uncertainty. First, the reported history of waste deposition practices and the waste drill core samples suggest that there is considerable anisotropy within the units, such that individual waste samples may not well represent the total contents of the impoundments or the landfills[1]. Second, observation of groundwater samples collected directly beneath some of the units showed considerable decrease in the concentration of constituents of concern compared with the waste leachate, suggesting that the *in situ* samples may not reflect all of the phenomena affecting the chemistry of material actually escaping the units. Third, comparison of the porewater samples with TCLP and SPLP samples prepared from the same drill core segments demonstrated that porewater concentrations were typically higher than the laboratory leachate samples for high concentration samples, and were lower for low-concentration (e.g. near detection limit) samples, but showed generally good agreement at moderate concentrations.

Issue: Waste Characteristics Data Were Very Limited for Comanaged Waste Landfills

EPA's database of comanaged waste characteristics included samples from only three landfills. Like the porewater samples collected at the surface impoundments in EPRI's comanagement site investigations, the landfill samples attempted to collect interstitial waters from cores drilled into the solid wastes. However, many of the samples contained insufficient free water for sample collection. EPRI therefore prepared 2:1 water-to-waste preparations that were allowed to equilibrate for a period of time prior to filtration and analysis. Comparison of the 2:1 extract data with the corresponding TCLP and SPLP data for four of these core samples indicated generally good agreement between the EPRI methodology and the TCLP and SPLP results. However, comparison of the landfill extract results with the concentrations observed in the impoundment porewater samples showed the landfill extracts to exhibit considerably lower contaminant concentrations than many of the porewaters from the impoundments.

[1] For example, none of the cores collected at the P4 site revealed pyrites, although pyrites were reportedly disposed intermittently within the unit since it began operation. Note, however, that sample representativeness is not an issue unique to *in situ* sampling.

EPA did not have sufficient information to conclude if the difference in observed concentrations between the landfill samples and the impoundment samples were representative of actual differences in the population, or were a result of the limited number of available samples. Faced with the paucity of data for comanaged wastes managed in landfills, EPA elected to combine the landfill and impoundment porewater samples into a single database to represent all comanaged wastes. The 95th percentile concentrations from the combined database were clearly dominated by impoundment sample concentrations.

7.2 SIMPLIFICATION OF MANAGEMENT PRACTICES AND ASSUMPTIONS AFFECTING LEACHING

Uncertainty regarding the contaminant concentration in leachate was discussed in Section 7.1. Uncertainties are also associated with factors affecting the total quantity of leachate available to the subsurface system. The issues relating to this general topic and discussed in this section are as follows:

- Simplification of actual surface impoundment construction
- Simplification of actual minefill project operation.
- Model assumptions regarding impoundment closure
- Uncertainty of infiltration rates
- Presence of liners at FFC waste management sites
- Simplification of actual landfill codisposal conditions
- Overprediction of actual contaminants available in impoundment and landfill scenarios

Issue: EPACMTP Impoundment May Oversimplify FFC Waste Impoundments

EPACMTP assumptions regarding impoundment design do not reflect the wide range of operating conditions observed in the population of comanagement sites. Specifically, EPACMTP assumes impoundments to exhibit a rectilinear plan view with uniform liner depth and uniform standing water depth. The ash/sediment layer is characterized by a uniform liner vertical hydraulic conductivity, and leachate flows only vertically across the ash sediment layer. In fact, many impoundments are valley-fill designs with widely varying ash deposition patterns. Ash layer thickness may vary significantly throughout the basin, along with standing water depth. Similarly,

different waste types may not be evenly distributed throughout a unit. For example, bottom ash and pyrites may be localized in a single corner of a unit while fly ash is deposited in another. Additionally, site-specific observations indicate ground-water mounding conditions exist at many impoundments and reflect the potential for horizontal as well as vertical release of leachate.

The magnitude and direction of error introduced into model results due to the restrictive definition of impoundment design can not be easily determined. In cases with very uneven distribution of ash layers, for example, infiltration rates may exceed those predicted, whereas the interaction with wastes (and so the mobilization of soluble constituents) may be below those predicted. The resulting effect of these competing influences on contaminant flux is unclear.

Issue: The Current Definition of Minefill Scenarios Do Not Capture the Range of Activities Conducted at Actual Sites

Minefill scenarios are characterized by three primary areas of uncertainty: minefill-project specific waste characterization data were unavailable; the incorporation of non-FFC wastes in fill materials as commonly practiced was not incorporated into scenario design, thereby overstating the contribution of FFC wastes to total leachate; and EPACMTP can not address fractured-flow conditions that may prevail in some mining-impacted areas.

EPA used comanaged waste characterization data and the FBC waste characterization data to characterize materials utilized in minefill projects. Since many FBC waste generators reported managing wastes in minefill projects, the application of these data to the minefill scenario was appropriate. However, EPA had comparatively little data on application of comanagement practices at minefill projects, and so could not determine the likelihood that such operators incorporate low-volume wastes with large-volume wastes prior to mine placement. Also, many minefill and mine reclamation projects were reported to incorporate local mine spoils into the total fill material. This observation indicated that the scenario both overestimated the volume of FFC wastes managed in identified projects and overstated the contribution of FFC waste leachate to the total leachate generated at such projects. The first condition may have resulted in an overstatement of the project area, a strong model driver. The second condition may have resulted in an overstatement of the starting constituent concentrations attributable to FFC wastes, also a

strong model driver. Accordingly, the waste and project area characterization data selected to define the minefill projects modeled may have significantly increased the estimated risk for the projects over those expected from scenarios capturing more realistic conditions.

It should also be noted that EPACMTP was not intended to model fractured flow conditions. Some minefill projects overlie areas of substantial underground mining disturbance. In some areas, ground-water movement may follow historic underground mine voids, as well as fractured host rock. Therefore, the results do not address the potential effects on mobility associated with such ground-water flow conditions.

Issue: EPACMTP Impoundment Does Not Account for Wastes Left In Place

The EPACMTP assumption that the impoundment leaching rate drops to zero following management unit closure significantly underestimates post-operational infiltration expected at most FFC waste management units. EPA found that many FFC waste management units, especially large utility comanaged waste impoundments, serve as the final resting place of the wastes, and are closed by allowing standing water to drain from the unit, applying a cover/cap, and then revegetating the surface. Such units will continue to generate leachate, albeit at a rate lower than that predicted during active operations.

EPA assumed that the EPACMTP impoundment scenario adequately reflected the risks associated with releases to the subsurface during the active life of the impoundment. Moreover, EPA assumed that the landfill scenario, as defined, adequately captured or bounded the risks associated with the post-operational period of wastes left in place after impoundment closure. However, the landfills and impoundments described in the scenarios represented distinct populations, with distinct sizes and locations, and with wastes in differing physical states (e.g. percent moisture content). Therefore, the reliability of the assumption that the landfill captures the risks of impoundments closed as landfills warranted additional scrutiny.

EPA compared the capacity of the landfill and impoundment distributions developed for the model scenarios. First, EPA found that the total capacity of the landfills and impoundments modeled were similar at the median and 95th percentile levels. Based on waste generation and

impoundment capacity information for the landfills, EPA found that roughly 60 percent of the impoundments were filled at closure. Accordingly, EPA concluded that the impoundments closed as landfills were generally smaller than or equal in size to the landfills modeled.

EPA also compared the ash depth and fill area at closure for the impoundments and the landfills, and found that landfills were roughly twice as deep and covered roughly 70-80 percent of the surface area of impoundments at both the median and 95th percentile levels. Area was demonstrated to be a sensitive model parameter for the landfill scenarios, suggesting that the landfill may somewhat underrepresent the leachate generation potential of the impoundments closed as landfills.

EPA compared the effects of location on the hydrogeological conditions prevailing at the landfills and impoundments. Specifically, EPA found little difference in the estimated median and 95th percentile values of unsaturated and saturated zone properties due to differences in the geographic distributions of the landfills and impoundments. However, EPA did find that the median recharge rate predicted by HELP modeling for the landfill locations was significantly lower than the recharge rate predicted for the surface impoundment locations at the median and 95th percentile levels. Because EPA assumed recharge and infiltration rates to be equal, the landfill scenario infiltration rates may have underrepresented the infiltration rates for impoundments closed as landfills. However, given the uncertainty in the application of the HWIR soil properties and the HELP model predictions to the FFC waste sites as performed in this model (see discussion in this Section), the finding of different recharge rates for the two populations requires additional investigation.

Finally, EPA considered the effect of leaching during the active phase of the impoundment on the availability of leachable metals in impoundments closed as landfills. As described in Section 4, the landfill scenario accounted for leaching behavior commencing at the closure of the landfill. Leaching reduced the remaining mobile metals until all metals were removed from the landfill. The total leachable metals were determined by the leachate starting concentration, an estimate of the ratio of leachate to total concentration, and the total quantity of wastes in the unit. The surface impoundment, however, significantly reduced the available leachable metals during its

active lifetime. Accordingly, both the starting leachate concentration and the remaining leachable metals for impoundments closed as landfills are expected to fall below those of the original landfill.

EPA did not quantitatively evaluate the aggregate impact of all of these factors on the relative potential for releases from landfills versus impoundments closed as landfills.

Issue: HWIR-Derived Infiltration Rates May Not Represent FFC Waste Infiltration Rates

Results in Section 5 showed that predicted risks from landfills exceeded risks from surface impoundments in deterministic scenarios. All infiltration rates used in modeling in this report were based on those generated by the HELP model and used for HWIR. The characteristics of fossil fuel combustion wastes, particularly FBC wastes, may be dissimilar to the characteristics assumed for the HWIR landfilled wastes and reflected in the available HELP model results.

The HELP model uses the following assumptions (and others) for development of the infiltration rates used in HWIR. A two foot soil cover was assumed to represent Subtitle D landfills without a liner. A cover crop of "fair" grass as cover material was assumed. Adequate recalculation of landfill infiltration rates requires sufficient knowledge of the waste's hydraulic conductivity, location assumptions, etc. Such recalculations were not performed, in part because of the high level of uncertainty or variability associated with these parameters in a nationwide analysis. However, assumptions regarding vegetation and soil properties affect infiltration rate. As one simple example, the universal soil loss equation predicts different runoff for different vegetation conditions. As another example, one of the EPRI site reports used the HELP model to calculate an infiltration rate based on several different closure scenarios and assumptions; each set of assumptions resulted in a different infiltration rate for the particular site. Infiltration rates were calculated at the AP site and presented in the site investigation report for seven closure assumptions; the infiltration rates ranged from 0.171 to 0.295 inches per year, reflective of the effect of closure options. The lowest value is 58 percent (0.171/0.295) of the highest value.

The change in infiltration rate calculated at the AP site was assumed to be proportional to the change that may occur at other sites, for purposes of this demonstration. Specifically, the

infiltration rate used in the deterministic analysis for assessing coal comanagement landfills was 0.0894 m/y. For this demonstration, the infiltration rate was reduced by multiplying the rate by 58 percent (derived above). Table 7-2 shows the effect reducing the landfill infiltration rate for selected constituents. As expected, a reduced infiltration rate increased the contaminant's dilution and attenuation between the source and the receptor well. The reduced infiltration rate, therefore, decreased risk at the receptor well.

Table 7-1. Effect of Infiltration Rate on Dilution and Attenuation Factor for Coal Comanagement Landfill Scenario

Constituent	Dilution and Attenuation Factor (DAF)	
	Original Conditions (Section 5 Deterministic Result	Reduced Infiltration Rate Conditions (Infiltration Rate Multiplied by 0.58)
Arsenic	3.17	16.46
Barium	1.76	2.58
Cadmium	23,708	273,530
Chromium VI	1.84	3.28
Nickel	1.77	17.7
Selenium	2.25	2.97
Note: 1. Chromium III, lead, mercury, vanadium and zinc produced extremely high DAFs for both runs and are not presented.		

These results show that changes in assumptions regarding infiltration rate in a landfill changes the DAF (and therefore the receptor well concentration). However, specific changes in this infiltration rate are not presented here, due in part to the variability in the hydraulic conductivity of the waste. This variability is discussed later in Appendix O.

Issue: Scenarios Do Not Account for Environmental Controls in Actual Waste Management Units

The risk assessment presented here addressed the potential risks to human health resulting from ground-water contamination from those remaining FFC waste management practices that EPA determined had the greatest potential to release constituents of concern to ground water. Each of the scenarios developed for the assessment was limited to waste management units with no environmental controls (e.g. liners, leachate collection). However, EPA assigned to the hypothetical waste management units the same hydrogeological, meteorological, size, and leachate characteristics observed or expected throughout the entire remaining waste universe (i.e. those actual units with and without environmental controls). In effect, EPA examined the potential risk from unlined management units that are of the same size and are located in the same

place as any actual unit, lined or unlined. In other words, EPA assumed that an unlined unit can be located anywhere that a lined unit could be found, and that an unlined unit could be the same size and could have the same waste characteristics as any lined unit.

EPA actually observed certain trends in waste management activities that reflect differing waste management decisions based on unit type, unit size, unit location, or other salient unit characteristics. For example, EPA observed that oil ash impoundments in Massachusetts were more likely to be lined than those in Florida. Accordingly, allowing the Monte Carlo analysis (or the calculation of median values for the deterministic analysis) to weight equally the likelihood of finding an unlined oil ash unit located in states such as Massachusetts may skew the results for states such as Florida.

EPA did not have a complete census of waste management units nationwide. Accordingly, EPA could not rule out the possibility of an unlined oil ash landfill located in Massachusetts. On the other hand, EPA did determine that Massachusetts represented one of the three highest States with regard to oil ash generation, and assumed that fact alone increased the likelihood of finding an unlined oil ash landfill in the State compared with other, lower oil-using States.

Issue: Commercial Landfill Scenarios May Overpredict Risk Due to Excessive Flux of Contaminants

The commercial landfill scenarios for oil combustion wastes and non-utility combustion wastes may overpredict the overall leachate concentration and underpredict the pulse duration for some metals. The net effect is to overstate risk by promoting an unrealistically high flux of constituents of concern into the subsurface. Both commercial scenarios were defined to represent the commingling of a quantity of FFC wastes of known leachate characteristics with a quantity of unspecified nonhazardous wastes of unknown (and inert) leachate characteristics in a commercial landfill setting. The scenarios were designed to identify the incremental risk associated with addition of FFC wastes to the commercial landfill. EPACMTP was developed to accommodate this scenario by allowing the explicit specification of the fraction of the total waste stream represented by the specified starting leachate concentration. However, the model appears to

apply a single starting leachate concentration to the entire quantity of infiltrate calculated based on the total landfill area, without adjustment for the fraction of wastes represented. While potentially unimportant for some metals (As, Cr(VI)), those metals with non-linear adsorption isotherms (Ba, Be, Cd, Cr(III), Cu, Pb, Ni, Ag, V, and Zn) may be ascribed an inflated starting concentration and therefore overpredict the potential down-gradient receptor well concentration.

EPA is currently exploring the sensitivity of the waste fraction variable, and has performed limited investigation of alternative methods of characterizing the commercial landfill scenario. Based on the findings to date, however, the results of the commercial landfill scenarios should be viewed with caution.

Issue: EPACMTP Overpredicts the Availability of Leachable Contaminants in Landfill and Impoundment Scenarios

EPACMTP calculates the total quantity of leachable metals contained within a landfill from the starting leachate concentration, the ratio of leachate to total concentration, and the total quantity of wastes contained within the landfill. The model then attributes the total quantity of the contaminant at a constant rate until it is gone. As a practical matter, the assumption of constant leachate concentration and the expectation that all metals within the waste matrix are available for leaching are potentially significant sources of error in the modeling of FFC wastes. Laboratory column leaching studies examining the leachability of constituents of concern in FFC wastes generally demonstrate reduction in leachable metals concentrations with successive pore volumes[2]. In fact, the leachability of a metal in FFC wastes depends in part on the volatility of the metal and its propensity to accumulate on ash surfaces or remain distributed throughout the particle matrix[3]. EPA did not identify information that quantifies the total available metals versus the total concentration of metals in FFC wastes.

[2] EPRI. 1986. "Mobilization and Attenuation of Trace Elements in an Artificially Weathered Fly Ash." Prepared by University of Alberta, Edmonton, Alberta, Canada.

[3] Eary, L.E., et. al. 1990. "Geochemical Factors Controlling the Mobilization of Inorganic Constituents from Fossil Fuel Combustion Residues: I. Review of the Minor Elements," in *Journal of Environmental Quality*, Vol. 19, April-June, 1990.

In the impoundment scenarios, EPACMTP does not calculate the total quantity of a constituent of concern. Rather, the flux of contaminant leaving the impoundment is described by the constant infiltration rate and the constant leachate rate. Therefore, the model can release to the subsurface more contaminant than the unit contains. In fact, EPA performed trial examinations of the total contaminant flux to determine whether mass was conserved by the model and found that, for some metals in the oil ash impoundment scenario, the total metals released from the management unit exceeded the total metals placed in the unit[4].

EPA did not quantify the magnitude of error that the availability and leaching rate assumptions may have contributed to the model outcomes. EPA notes, however, that since dilution and attenuation are dynamic processes that depend in part on the rate of contaminant flux relative to the replacement rate of ground water, those assumptions that overstate the rate of flux of contaminants inflate the estimated risks.

7.3 SIMPLIFICATION OF PROCESSES AFFECTING FATE AND TRANSPORT

Once the contaminants leave the management unit and enter the subsurface, there is additional uncertainty in modeling the transport of these contaminants and assessing risk to a downgradient receptor. Specifically, the effect of the following parameters on simulating fate and transport are discussed here:

- Simplifying the variability of actual hydrogeologic factors
- FFC waste management facilities may be located in karst conditions
- Receptor well location influences potential risk
- Speciation of chromium in ground-water

Issue: Site-specific Environmental Setting Data Were Limited

Application of meteorological and hydrogeological information from HWIR to sites based on identification of States alone may result in inaccurate characterization of conditions prevailing at FFC waste sites. EPA did not obtain sufficient site-specific information to adequately represent

[4] EPA did not perform this analysis for the current scenarios. See Appendix K for a review of sensitivity analyses generally.

the variability of conditions expected to obtain for any of the remaining FFC waste sectors modeled. EPA attempted to address this by identifying within the HWIR data the probable meteorologic and hydrogeologic conditions describing sites within each State. State-specific data were then ascribed to each FFC waste site. EPA assumed that the HWIR distribution of sites captured the variability of conditions within each State, and that FFC waste sites would be distributed within any given State similarly to the HWIR sites, with respect to the variability of conditions. EPA believed that this approach would allow consideration of the wide range of conditions observed nationwide, without juxtaposing unlikely combinations of meteorology and hydrogeology (e.g. West Virginia rainfall patterns with West Texas hydrogeological conditions).

The uncertainty inherent in this approach varied for the non-utility waste sector. For example, the HWIR population may be expected to represent very well the non-utility population, since the HWIR sites were generally large industrial sites in the same industry categories as those most represented within the non-utility fossil fuel combustor universe. However, utility waste sites may reflect subtle but significant differences in geographic and hence hydrogeologic distribution. As an example, many or most utilities are located near large surface water bodies, increasing the potential that the sites may be underlain by shallow ground-water compared with assigned values. EPA compared the median values of the HWIR-derived distributions with utility industry-specific information (e.g. EPRI 1984) to confirm that the HWIR values fell within the range of independently-derived values. Similarly, EPA confirmed that the site-specific observations in the EPRI comanagement reports fell within the range of HWIR-derived values. Neither of these efforts, however, ensured that the relative prevalence of the given HWIR conditions were appropriate for the utility universe.

The magnitude and direction of error resulting from extrapolation of the HWIR environmental settings data to the individual remaining waste categories based only on State information only could not be determined. However, EPA concluded that the use of these data on a State by State basis reduced the error associated with random selection of environmental settings information from nation-wide distributions. Further, EPA concluded that the chosen approach provided a reasonable reflection of region-specific environmental conditions where needed (e.g. oil-fired utilities).

Issue: Some FFC Sites May Be Located in Karst Terrane

EPACMTP was not developed for, and can not simulate, fractured flow conditions such as may prevail in areas of karst hydrogeology or heavily disturbed mining areas underlain with mine workings. EPA did not complete a systematic review of the occurrence of FFC waste management sites in areas of fractured flow conditions. EPA noted, however, that at least one of the detailed site investigations performed by EPRI focused on a site developed in karst terrain. Further, EPA found that some of the minefill projects for which information were available were developed in areas where ground-water flow was controlled by the presence of underground mine workings. Therefore, the results of this risk assessment are silent with respect to an unknown portion of the FFC waste universe located in areas dominated by channel flow conditions.

Issue: Selection of receptor well location

The receptor well was assumed to be 150 meters downgradient, on plume centerline, for all high end deterministic scenarios. This assumption, together with the study period, significantly affect the calculated risk. Analysis of the time for a constituent to reach a carcinogenic risk of $1x10^{-6}$ or a noncarcinogenic hazard quotient of 1 demonstrates that most constituents do not reach concentrations of concern for thousands of years, despite a receptor well location of only 150 meters from the management unit. Accordingly, small changes in the distance to the receptor well would cause a significant decrease in receptor well concentration predicted at the same time. The long study period introduces additional uncertainties in the modeling assumptions regarding the infiltration rate and leachate concentrations over this period of time as discussed earlier in this section.

Issue: Chromium Speciation

Chromium is present in FFC wastes at several sites. The species of chromium is an important consideration from a toxicity standpoint. Specifically, the RfD for chromium III is 200 times greater than the RfD for chromium VI (based on IRIS). This means that if the chromium in FFC wastes is in the trivalent form, it would be much less toxic than if the constituent were present in its hexavalent form.

The deterministic high end analyses in Section 5 showed a risk from this constituent (HQ

greater than 1) in both co-management and non-utility waste scenarios. Both of these scenarios use the same set of comanagement data from EPRI. Although EPRI did not provide measurements of chromium speciation in their sampling results, they did provide data for other analytes, notably pH and redox potential, so that the predominant species can be inferred. A complete analysis is presented in Appendix H. Based on this analysis, chromium III is the predominant species in FFC comanaged wastes.

This finding results in the diminishing of risk from chromium, if the following two assumptions are valid: (1) the species does not change from chromium III to chromium VI in the subsurface prior to the receptor well, and (2) the speciation of chromium in comanaged wastes is reflective of the speciation of chromium in non-utility wastes. Even with the uncertainties in these assumptions, the risks from chromium presented in Section 5 are overestimates because they assume all of the chromium is present in the hexavalent state.

7.4 DOSE-RESPONSE UNCERTAINTY

Of the contaminants evaluated in this analysis, two (arsenic and beryllium) are carcinogens. Section 5 demonstrates that arsenic shows high potential risk for all scenarios evaluated. Therefore, uncertainties in the derivation and application of dose-response factors in general and in the behavior of arsenic in particular will have a significant effect on results.

The cancer slope factor used in this analysis is from EPA's Integrated Risk Information System (IRIS). The carcinogenicity assessments in IRIS begin with a qualitative weight-of-evidence judgment as to the likelihood that a chemical may be a carcinogen for humans. This judgment is made independent of consideration of the agent's potency. A quantitative assessment, which may include an oral slope factor and oral and/or inhalation unit risks, is then presented. The oral slope factor is an upper-bound estimate of the human cancer risk per milligram of agent per kilogram of body weight per day.

In general, IRIS values cannot be used to accurately predict the incidence of human disease or the type of effects that chemical exposures have on humans.

This is due to the numerous uncertainties involved in risk assessment, including those associated with extrapolations from animal data to humans and from high experimental doses to lower environmental exposures.

The organs affected and the type of adverse effect resulting from chemical exposure may differ between study animals and humans. In addition, many factors besides exposure to a chemical influence the occurrence and extent of human disease.

The latest IRIS update regarding arsenic is April 1998. Conclusions from an EPA sponsored Expert Panel on Arsenic Carcinogenicity from May 1997 are provided. The Expert Panel believed that, "it is clear from epidemiological studies that arsenic is a human carcinogen via the oral and inhalation routes." They also concluded "that one important mode of action is unlikely to be operative for arsenic". The panel agreed that arsenic and its metabolites do not appear to directly interact with DNA." In addition, the panel agreed that, "for each of the modes of action regarded as plausible, the dose-response would either show a threshold or would be nonlinear". The panel agreed, however, "that the dose-response for arsenic at low doses would likely be truly nonlinear, i.e., with a decreasing slope as the dose decreased. However, at very low doses such a curve might be linear but with a very shallow slope, probably indistinguishable from a threshold.".

The cancer slope factor for arsenic currently listed in IRIS (and used in this report) is calculated by the multistage model which provides a linear estimation of risk (i.e, the cancer slope factor) at low concentrations. One uncertainty of applying the multistage model to the dose-response of arsenic, therefore, is in the assumption of linear response to risk at these concentrations.

8. CONCLUSIONS

In this study of the risks to human health from ground water potentially affected by fossil fuel combustion (FFC) wastes, EPA found that the remaining waste universe represents a large and diverse population of waste management units located throughout the Nation. To determine the potential risks from this diverse population, EPA developed a risk assessment methodology that considered each of four remaining waste categories independently. These categories included coal-fired utility comanaged wastes, oil-fired utility wastes, fluidized bed combustion (FBC) wastes, and coal-fired non-utility wastes. EPA studied the characteristics of remaining wastes and determined metals to be the primary constituents of concern. EPA also focused its study on those management practices believed to present the greatest potential for release of contaminants to ground water: surface impoundments, landfills, and minefills.

EPA conservatively estimated the high-end concentration of each constituent of concern for each sector by using the 95th percentile concentration from the limited data available for each sector. In its screening assessment, EPA compared these concentrations directly with health-based benchmark values derived to represent the threshold risk concentration for an adult resident receptor. The concentrations of most of the metals of concern exceeded their respective benchmark values, so the screening assessment did not eliminate many metals from further consideration. Arsenic, cadmium, chromium, lead, nickel[1], selenium, and vanadium remained of concern for all sectors.

The screening assessment addressed the exposure of the adult resident to undiluted leachate. This estimate was used to target specific constituents for modeling using EPACMTP. EPA estimated the maximum concentration expected to occur in a near-by well (150 meters) for each of ten remaining waste management scenarios. To ensure a protective estimate, EPA identified the two most sensitive model parameters (concentration and well location) and set these to their high-end values in all of the scenarios modeled deterministically. EPA then corroborated

[1] Nickel was not of concern for FBC wastes, based on screening.

the relative conservatism of the deterministic models by performing probabilistic analyses of the same waste management scenarios.

The following subsections present EPA's preliminary conclusions for each of the waste categories, followed by brief overall conclusions.

8.1 COAL-FIRED UTILITY COMANAGED WASTES

Arsenic presented the highest potential for risk from comanaged wastes in all waste management scenarios. In addition, chromium, nickel, and selenium showed small (<5x) exceedences of their risk thresholds in downgradient ground water for the landfill and minefill scenarios. The time at which ground-water concentrations reached the benchmark concentration was 1,500 years or more for all constituents and scenarios except for arsenic in the surface impoundment (500 years).

EPA found that modeling uncertainty and error may have led to substantial overestimation of risks. First, calculation of the 95th percentile from the small sample size may have given excessive weight to the worst-case observations of arsenic concentration at one facility. Samples from that site demonstrated the local influence of acid generation from pyrite oxidation on porewater chemistry, but may not well represent the chemistry of leachate migrating from any actual unit. Second, EPACMTP may overstate the rate of leachate generation for both impoundments and landfills (this was true for all scenarios) and may maintain excessive leachate concentration throughout the modeling period, both of which would accelerate contaminant flux to the subsurface and inflate the peak downgradient concentration.

EPA found that chromium is not expected to pose actual risk because of the evidence showing the predominance of trivalent chromium species over hexavalent chromium species in these management units. Thus, chromium was concluded not to be of concern.

Comparison of the high-end results with the probabilistic results for nickel and selenium showed the high-end results to correspond to the 99th percentile Monte Carlo result, and that the

95th percentile Monte Carlo result in both cases fell below the threshold level of risk. This led EPA to conclude that nickel and selenium also should be dismissed from concern.

In summary, for this waste sector, EPA concluded that potential risks from arsenic warranted additional attention. Of particular interest are the accuracy with which leachate infiltration rate and concentration are calculated throughout the modeling period. EPA also concluded that the quantitative measure of risk associated with the high-end scenario should be viewed with caution pending completion of additional review.

8.2 OIL-FIRED UTILITY WASTES

Arsenic, nickel, and vanadium presented the highest potential for risks from oil-fired utility wastes. Each was predicted to exceed the risk threshold in down-gradient ground water in all scenarios. Vanadium and nickel were predicted to exceed their HBL concentrations in a near-by receptor well in 50 years or less for the oil ash surface impoundment. Arsenic exceeded the benchmark in the receptor well after 400 years for the surface impoundment and 2,800 years for the landfill.

Again, EPA found that uncertainty and modeling error may have overestimated the risks associated with oil-fired utility waste management. Of principle importance, EPACMTP appeared to overstate the rate of leachate generation for both impoundments and landfills and may have maintained excessive leachate concentration throughout the modeling period, both of which would accelerate contaminant flux to the subsurface and inflate the peak downgradient concentration.

EPA concluded that potential risks from arsenic, nickel, and vanadium from oil-fired utility wastes warranted additional attention. Of particular interest is the accuracy with which leachate generation rate and concentration are calculated throughout the modeling period.

8.3 FLUIDIZED BED COMBUSTION (FBC) WASTES

Arsenic demonstrated the highest potential for risk from FBC wastes[2]. The predicted downgradient drinking water concentration of arsenic exceeded the CSF-based benchmark concentration for the landfill and the minefill scenarios after about 3,600 years. Risks from the minefill and the landfill were predicted to be similar. Antimony also demonstrated potential risk, exceeding the risk threshold by a factor of 20 in receptor well water. However, this result was at the 99th percentile of the Monte Carlo results; the exceedence at the 95th percentile was by a factor of only 1.7. Because of the conservatism of the assumptions and the relatively low exceedence, antimony should not be of significant concern.

As with the other waste types, EPA found that uncertainty and modeling error may have overestimated the risks associated with FBC wastes. The potential for these wastes to exhibit "self-cementing" properties leading to very low hydraulic conductivity was not accounted for in the model. In addition, EPACMTP may have overestimated leachate generation rate and leachate concentration. EPA also found that the modeling scenario employed did not account for the presence of other materials in FBC waste minefills.

In summary, EPA concluded that the risks from arsenic from FBC wastes warranted additional attention. Of particular interest are accounting for the hydraulic conductivity of the wastes in modeling, and verifying the accuracy with which the model calculates infiltration rate and leachate concentration.

[2] Beryllium also demonstrated a high potential for risk from FBC wastes. Ground-water concentrations of Be were predicted to exceed the CSF-based HBL by a factor of 1,000 or more in wells located 150 meters from a hypothetical FBC waste landfill. However, since completing these analyses, EPA has vacated the CSF upon which these data are based. Appendix O shows the impact of considering alternative benchmarks in evaluating risks from Be.

8.4 NON-UTILITY FFC WASTES

Arsenic presented the highest potential risks for coal-fired non-utility wastes. EPA found this sector to present the highest uncertainty, however. First, because no data were available to characterize the wastes from this sector, EPA assumed the characteristics of coal-fired utility comanaged wastes to represent the non-utility wastes. Accordingly, results from this sector were driven by the same samples and sample size considerations as those for the comanaged wastes. Further, the landfill model exhibited the same potential to overstate the infiltration rate and leachate concentration over time.

As was the case for coal-fired utility comanaged wastes, EPA concluded that arsenic from non-utility fossil fuel combustion wastes may warrant additional attention, pending the resolution of modeling issues relating to this and other scenarios.

8.5 SUMMARY

Most metals did not demonstrate appreciable risk in the high-end modeling assessment. However, EPA found that one or more metals exceeded the threshold risk concentration in the near-by well sometime within 10,000 years for all scenarios considered. Barium, cadmium, copper, lead, mercury, thallium, and zinc were not predicted to exceed threshold risk concentrations in any of the on-site management units[3]. In contrast, EPA found that arsenic exceeded the threshold of concern for all scenarios, and that other metals (antimony, chromium, nickel, selenium, and vanadium) appeared above benchmark levels for one or more scenarios.

Upon consideration of all factors, EPA concluded that arsenic is the primary constituent of concern from FFCs. In addition, for oil combustion wastes, EPA concluded that vanadium and nickel warrant further consideration.

[3] Barium and cadmium each exceeded benchmark values at peak concentrations in commercial landfill scenarios. However, each high-end result was found to exceed the 95th percentile Monte Carlo result. Moreover, EPA has found the results of the commercial landfill scenarios to be problematic, as discussed in Section 7.

For most constituents evaluated, risks to young children (less than ten years old) were approximately twice as great as risks to adults. However, the increased risk relative to adults did not reveal any new constituents of unique concern for child receptors.

Modeled risks were generally found to be higher for landfills than for surface impoundments. Since the landfill scenario was intended to capture the risks associated with impoundments closed in place, the over-all risks associated with the two scenarios could be expected to be similar. Moreover, risks were generally found to be similar between landfills and minefills. Deterministic modeling results suggested potentially higher risks for off-site management of wastes compared with on-site practices. However, results for off-site and minefill scenarios were very preliminary and require additional analysis to evaluate the accuracy and significance of preliminary findings.

Coal-fired utility comanaged wastes showed the highest potential risk from arsenic for all three management scenarios, compared with other wastes. Oil ash and FBC wastes showed similar risks from arsenic. Non-utility arsenic risks followed those of the comanaged wastes based on the assumption that waste characterization between the sectors would be similar.

Overall, EPA found that the Monte Carlo assessment supported the conclusion that the deterministic scenarios were sufficiently conservative to represent high-end risks. Analysis of uncertainty further supported the conclusion that the model results are very conservative in predicting long-term contaminant migration. Finally, EPA concluded that data limitations and model performance issues present sufficient uncertainty that the quantitative evaluation of risk potential should be viewed with caution.

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APPENDIX A

MODEL INPUTS

Table A-1. EPACMTP Model Inputs: Coal-Fired Utility Comanaged Waste Surface Impoundment Scenario (Scenario CS)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
Source-Specific Variables			
AREA, management unit area	CT: 0.364×10^6 m^2 (90 acres) HE: 1.67×10^6 m^2 (412 acres)	EPRI Comanagement database: 50th and 95th percentile value for surface impoundments regardless of comanagement status.	Comanagement survey is more recent than other sources, and more representative of comanagement units.
CZERO, leachate concentration	Constituent dependent	EPRI site investigations.	
RECHRG, recharge rate	CT: 0.3256 m/y	Based on HWIR distribution for silty loam soil. Assigned 1 of 97 climatic centers to each of the approximately 100 impoundments in the comanagement data file (based on its location), and calculated overall statistics.	The HWIR distributions are used because they are available and representative of the United States, and no other source directly provides this information. Uses HELP-generated rate.
SINFIL, infiltration rate from unit	Derived (m/y)	None	Direct information on infiltration rates, from the site investigations, are limited to five sites. The alternative calculation method treats this variable as derived from liner conductivity, thickness, and depth. These parameters are available from multiple sites and allows the model to account for the uncertainty.
HZERO, ponding depth of surface impoundment	CT: 1.8 m HE: 19 m	EPRI Comanagement survey: 5th and 50th percentile values derived/estimated from 20-year waste generation, area, and capacity. Based on annual waste generation rates and an assumed lifetime of 40 years.	Manipulation of survey data represents more sites than data obtained from site investigations.
DLINR, liner thickness, m	CT: 3.4 m HE: 0.43 m	Comanagement survey: 5th and 50th percentile values derived/estimated from 20-year waste generation, area, and capacity.	Manipulation of survey data represents more sites than data obtained from site investigations.

Table A-1. EPACMTP Model Inputs: Coal-Fired Utility Comanaged Waste Surface Impoundment Scenario (Scenario CS) (Continued)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
CLINR, hydraulic conductivity of liner	CT: 0.315 m/y	Central tendency value corresponds to 1E-06 cm/s.	Conductivity varies between waste types; additionally, data are available only sporadically from site investigations. Large uncertainty is associated with this parameter.
TSOURC, duration of leaching	Constant: 40 yrs	Professional judgment	Assumed operational life based on input from EPRI.
Metals-Specific Variables			
METAL_ID	Constituent dependent	EPACMTP Metals Background Document	Only metals with isotherms are used in modeling.
USPH, soil and aquifer pH	CT: 6.92 HE: 4.73 or 9.02	EPRI site investigations: downgradient and upgradient ground water samples (E and F samples).	Measured values from downgradient ground water monitoring wells are preferred to 1984 EPRI report. Unsure if high end represents 5th or 95th percentile case.
FEOX, iron hydroxide concentration in soil and aquifer	CT: 0.562 % HE: 0.0675 or 1.057 %	HWIR Distribution	HWIR is only source of data. CT value based on 50th percentile, high end based on 5th and 95th percentiles.
LOM, concentration of dissolved organic carbon in the waste leachate	CT: 9.49 mg/L HE: 1.44 or 181 mg/L	EPRI site investigations: water leaving the units and entering the aquifer (C and D samples)	Measured values from FFC sites are preferred to the HWIR distributions because HWIR uses data from MSWs, which do not represent FFC sites.

Table A-1. EPACMTP Model Inputs: Coal-Fired Utility Comanaged Waste Surface Impoundment Scenario (Scenario CS) (Continued)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
USNOM, unsaturated zone percentage organic matter (should be same as POM)	CT: 1.58 (%) HE: 0.35 or 4.50 (%)	1984 EPRI Report (Table 3-30).	1984 Report preferable to HWIR because more relevant to FFC sites.
ASNOM, aquifer fraction organic carbon (should be same as FOC)	CT: 0.032 HE: 0.061 or 0.003	HWIR	1984 Report data likely represents unsaturated zone only; HWIR presents only other source.
Unsaturated Zone Variables			
Saturated conductivity	Constant: 0.343 cm/hr	HWIR (mean value for silt loam)	Silt loam is most common soil type.
α moisture retention parameter	Constant: 0.019 cm^{-1}	HWIR (mean value for silt loam)	Silt loam is most common soil type.
β moisture retention parameter	Constant: 1.409	HWIR (mean value for silt loam)	Silt loam is most common soil type.
Res. Water content	Constant: 0.068	HWIR (mean value for silt loam)	Silt loam is most common soil type.
Sat. Water content	Constant: 0.45	HWIR (mean value for silt loam)	Silt loam is most common soil type.

Table A-1. EPACMTP Model Inputs: Coal-Fired Utility Comanaged Waste Surface Impoundment Scenario (Scenario CS) (Continued)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
DSOIL, thickness of unsaturated zone	CT: 8.3 m HE: 0 m	Central tendency depth of unsaturated zone is the difference of average unit depth and average depth to water table. Average depth to water table at coal combustion sites is calculated from 1984 EPRI Report, while average impoundment depth is calculated from comanagement survey. High-end value is from EPRI site investigation reports.	These industry specific data are preferred to HWIR (not industry-specific) or only site investigations (fewer sites represented).
Dispersivity	Derived	HWIR	
% organic matter	CT: 1.58 (%) HE: 0.35 or 4.50 (%)	1984 EPRI Report; see USNOM variable.	Values are higher than recorded for HWIR.
Bulk density	CT: 1.42 g/cm^3 HE: 1.85 or 0.89 g/cm^3	1984 EPRI Report: 5th, 50th, and 95th percentiles provided.	More utility-specific than nationwide data from HWIR.
Saturated Zone Parameters			
DIAM, average particle diameter in aquifer	CT: 0.021 cm HE: 8.9e-04 or 0.23 cm	HWIR: 5th, 50th, and 95th percentile value	Data are available in EPRI reports for determining ground-water velocity. However, variability is very broad, both within and between sites. Therefore insufficient data are available for determining a reasonable distribution.
POR, aquifer porosity	CT: 0.41 HE: 0.32 or 0.53	HWIR	
BULKD, aquifer bulk density	CT: 1.56 g/cm^3 HE: 1.25 or 1.80	HWIR	

Table A-1. EPACMTP Model Inputs: Coal-Fired Utility Comanaged Waste Surface Impoundment Scenario (Scenario CS) (Continued)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
ZB, aquifer saturated thickness	CT: 15.20 m	Based on HWIR and weighted to surveyed population. Each impoundment in the EPRI co-management data file was assigned to one of 13 hydrogeologic zones based on the predominant subsurface conditions of the meteorological zone.	
XKX, longitudinal hydraulic conductivity, K_x	CT: 315 m/y	Based on HWIR and weighted to surveyed population. Same basis as variable ZB (aquifer thickness).	
Anistropy ratio	1	Assumed	Values greater than 1 have been observed in limited EPRI site investigations; however, data are limited. This value is a default HWIR assumption.
GRADNT, hydraulic gradient	CT: 0.009	Based on HWIR and weighted to surveyed population. Same basis as variable ZB (aquifer thickness).	
VXCS, regional groundwater seepage velocity	derived (m/y)		
AL, longitudinal dispersivity	CT: 4.64 m HE: 0.32 or 68 m	HWIR distribution: 5th, 50th, and 95th percentiles	
AT, transverse dispersivity ratio	Constant: 8	HWIR recommendation	

Table A-1. EPACMTP Model Inputs: Coal-Fired Utility Comanaged Waste Surface Impoundment Scenario (Scenario CS) (Continued)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
AV, vertical dispersivity ratio	Constant: 160	HWIR recommendation	
TEMP, temperature of ambient aquifer water	CT: 17.5 °C	Based on HWIR and weighted to surveyed population. Same basis as variable ZB (aquifer thickness).	
PH, ambient ground-water pH	CT: 6.92 HE: 4.73or 9.02	EPRI site investigations. See USPH variable.	
FOC, fraction organic carbon	CT: 0.032 HE: 0.061 or 0.003	HWIR. See ASNOM variable.	
Receptor Well Location	HE: 150 meters, on centerline, depth is at water table	Assumed value.	Well location is specified in three coordinates. This placement reflects a high-end parameter.

Table A-2. EPACMTP Model Inputs: Coal-Fired Utility Comanaged Waste Landfill Scenario (Scenario CL)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
Source-Specific Variables			
AREA, management unit area	CT: 0.267×10^6 m^2 (66 acres) HE: 1.33×10^6 m^2 (328 acres)	EPRI Comanagement database: 50th and 95th percentile value for landfills regardless of comanagement status.	Comanagement survey is more recent than other sources, and more representative of comanagement units.
CZERO, leachate concentration	Constituent dependent	EPRI site investigations.	Same data as for coal comanagement impoundment scenario.
Cw/Cl value (waste to leachate concentration)	Constituent dependent	EPRI site investigations.	Waste-to-leachate concentrations were calculated for paired data sets (i.e., measurements of total and pore waste concentrations). A median Cw/Cl value was determined from this array.
RECHRG, recharge rate	CT: 0.0894 m/y	Based on HWIR distribution for silty loam soil. Assigned 1 of 97 climatic centers to each of the approximately 100 landfills in the comanagement data file (based on its location), and calculated overall statistics.	Same as coal comanagement impoundments.
SINFIL, infiltration rate from unit	CT: 0.0894 m/y	Same as recharge rate. Infiltration rate should equal recharge rate (model default).	Infiltration rate estimation can be improved by comparing HELP-model assumptions to actual FFC waste management conditions.
DEPTH, depth of landfill	CT: 9.45 m HE: 33.53 m	Comanagement survey: 50th and 95th percentile values derived/estimated from area and capacity.	Manipulation of survey data represents more sites than data obtained from site investigations.

Table A-2. EPACMTP Model Inputs: Coal-Fired Utility Comanaged Waste Landfill Scenario (Scenario CL) (Continued)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
waste fraction	Constant: 1	EPRI reports.	Assume that comanaged wastes of concern are only materials disposed in landfill.
waste density	CT: 1.19 g/cm^3	Assumed	Corresponds to 1 ton=1 cubic yard. Slightly lower than densities in 1988 Report to Congress for dry ash (wet ash is expected to be less dense).
TSOURC, duration of leaching	Derived	Model defalt	Model default; assumes all contaminants will leach out.
Metals-Specific Variables: Same as Scenario CS (Table A-1)			
Unsaturated Zone Variables: Same as Scenario CS (Table A-1)			
Saturated Zone Parameters: Same as Scenario CS (Table A-1), except as noted below			
ZB, aquifer saturated thickness	CT: 15.20 m	Based on HWIR and weighted to surveyed population. Each of the approximately 100 landfills in the EPRI co-management data file was assigned to one of 13 hydrogeologic zones based on the predominant subsurface conditions of the meteorological zone.	
XKX, longitudinal hydraulic conductivity, K_x	CT: 315 m/y	Based on HWIR and weighted to surveyed population. Same basis as variable ZB (aquifer thickness).	
GRADNT, hydraulic gradient	CT: 0.009	Based on HWIR and weighted to surveyed population. Same basis as variable ZB (aquifer thickness).	

Table A-2. EPACMTP Model Inputs: Coal-Fired Utility Comanaged Waste Landfill Scenario (Scenario CL) (Continued)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
TEMP, temperature of ambient aquifer water	CT: 12.5 °C	Based on HWIR and weighted to surveyed population. Same basis as variable ZB (aquifer thickness).	

Table A-3. EPACMTP Model Inputs: Minefill Scenarios for Coal-Fired Utility Comanaged Wastes (Scenario CF) and FBC Wastes (Scenario FF)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
Source-Specific Variables			
AREA, management unit area	CT: 141,000 m^2	Calculated from Pennsylvania minefill projects data.	Assumed to represent commercial offsite landfill.
CZERO, leachate concentration	Constituent specific	Same as for other FBC or coal co-management scenarios.	
CZERO, leachate concentration	Constituent specific.	Same as for other FBC or coal co-management scenarios.	
RECHRG, recharge rate	CT: 0.0789 m/y	Based on HWIR distribution for silty loam soil. Assigned 1 of 97 climatic centers to each of the 8 states most likely to have minefill projects (WY, IL, IN, KY, OH, WV, PA, MD) and calculated overall statistics.	
SINFIL, infiltration rate from unit	CT: 0.0789 m/y	Same as recharge rate. Infiltration rate should equal recharge rate (model default).	Infiltration rate estimation can be improved by comparing HELP-model assumptions to actual FFC waste management conditions.
DEPTH, depth of landfill	CT: 7.56 m	The Pennsylvania data provided capacity and area. Depth is calculated as capacity/area for each project; median depth was determined from all projects.	
waste fraction	CT: 100 %	Assumes no other disposal of other materials.	

Table A-3. EPACMTP Model Inputs: Minefill Scenarios for Coal-Fired Utility Comanaged Wastes (Scenario CF) and FBC Wastes (Scenario FF) (Continued)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
waste density	CT: 1.19 g/cm^3	Assumed.	Same as coal comanagement landfill scenario.
TSOURC, duration of leaching	Derived	Model default.	Model default; assumes all contaminants will leach out.
Metals-Specific Variables: Same as Scenario NM (Table A-8)			
Unsaturated Zone Variables: Same as Scenario NM (Table A-8), except as noted below			
DSOIL, thickness of unsaturated zone	CT: 6.1 m	Based on HWIR and weighted to surveyed population. Same basis as variable ZB (aquifer thickness).	
Saturated Zone Parameters: Same as Scenario NM (Table A-8), except as noted below			
ZB, aquifer saturated thickness	CT: 15.20 m	Based on HWIR and weighted to surveyed population. Each of the 12 relevant generators in the oil ash report was assigned to one of 13 hydrogeologic zones based on the predominant subsurface conditions of the meteorological zone. Assumes landfill is proximate to facility.	
XKX, longitudinal hydraulic conductivity, K_x	CT: 300 m/y	Based on HWIR and weighted to surveyed population. Same basis as variable ZB (aquifer thickness).	
GRADNT, hydraulic gradient	CT: 0.009	Based on HWIR and weighted to surveyed population. Same basis as variable ZB (aquifer thickness).	

Table A-3. EPACMTP Model Inputs: Minefill Scenarios for Coal-Fired Utility Comanaged Wastes (Scenario CF) and FBC Wastes (Scenario FF) (Continued)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
TEMP, temperature of ambient aquifer water	CT: 12.5 °C	Based on HWIR and weighted to surveyed population. Same basis as variable ZB (aquifer thickness).	

Table A-4. EPACMTP Model Inputs: Oil-Fired Utility Waste Surface Impoundment Scenario (Scenario OS)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
Source-Specific Variables			
AREA, management unit area	CT: 3,600 m^2 (0.90 acres) HE: 8,900 m^2 (2.2 acres)	EPRI Oil Ash Report.	Based on statistics from 9 facilities.
CZERO, leachate concentration	Constituent dependent	Oil ash data file, both TCLP and EP values.	
RECHRG, recharge rate	CT: 0.1016 m/y	Based on HWIR distribution for silty loam soil. Assigned 1 of 97 climatic centers to each of the nine impoundments in the oil ash report (based on its location), and calculated overall statistics.	The HWIR distributions are used because they are available and representative of the United States, and no other source directly provides this information.
SINFIL, infiltration rate from unit	Derived (m/y)	None.	Same as coal combustion comanagement scenario.
HZERO, ponding depth of surface impoundment	CT: 1.17 m HE: 2.6 m	EPRI Oil Ash Report. Assumes periodic dredging.	Calculated from solids and wastewater throughput from nine facilities. Median of nine facilities.
DLINR, liner thickness, m	CT: 0.21 m HE: 0.098 m	EPRI Oil Ash Report. Assumes periodic dredging.	Calculated from solids and wastewater throughput from nine facilities. Median of nine facilities.
CLINR, hydraulic conductivity of liner	CT: 0.315	Same as coal comanagement surface impoundment.	Same as coal comanagement surface impoundment.

Table A-4. EPACMTP Model Inputs: Oil-Fired Utility Waste Surface Impoundment Scenario (Scenario OS) (Continued)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
TSOURC, duration of leaching	Constant: 40 yrs	Professional judgment.	Assumed operational life based on input from EPRI.
Metals-Specific Variables: Same as Scenario CS (Table A-1)			
Unsaturated Zone Variables: Same as Scenario CS (Table A-1), except as noted below			
DSOIL, thickness of unsaturated zone	CT: 6.98 m HE: 2.2 m	Central tendency depth of unsaturated zone is the difference of average unit depth and average depth to water table. Average depth to water table at oil combustion sites is calculated from 1984 EPRI report (based on data limited to east coast facilities), while average impoundment depth is calculated from co-management survey. High-end depth of unsaturated zone is the difference of average unit depth and minimum depth to water table.	These industry-specific data are preferred to HWIR (not industry-specific).
Saturated Zone Parameters: Same as Scenario CS (Table A-1), except as noted below			
ZB, aquifer saturated thickness	CT: 15.20 m	Based on HWIR and weighted to surveyed population. Each of the 9 relevant impoundments in the oil ash report comanagement data file was assigned to 1 of 13 hydrogeologic zones based on the predominant subsurface conditions of the meteorological zone.	
XKX, longitudinal hydraulic conductivity, K_x	CT: 315 m/y	Based on HWIR and weighted to surveyed population. Same basis as variable ZB (aquifer thickness).	
GRADNT, hydraulic gradient	CT: 0.009	Based on HWIR and weighted to surveyed population. Same basis as variable ZB (aquifer thickness).	

Table A-4. EPACMTP Model Inputs: Oil-Fired Utility Waste Surface Impoundment Scenario (Scenario OS) (Continued)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
TEMP, temperature of ambient aquifer water	CT: 22.5 °C	Based on HWIR and weighted to surveyed population. Same basis as variable ZB (aquifer thickness).	

Table A-5. EPACMTP Model Inputs: Oil-Fired Utility Waste Onsite Monofill Scenario (Scenario OM)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
Source-Specific Variables			
AREA, management unit area	CT: 4,860 m^2	Based on median ash generation rate from EPRI Oil Ash report. Area and depth are calculated from this volume.	
CZERO, leachate concentration	Constituent specific	Same as impoundment scenario (Scenario OS, Table A-4).	
Cw/Cl value (waste to leachate concentration)	Constituent dependent	Same as oil co-disposal scenario (Scenario OL, Table A-6).	
RECHRG, recharge rate	CT: 0.1016 m/y	Based on HWIR distribution for silty loam soil. Same as oil combustion waste co-disposal landfill scenario.	
SINFIL, infiltration rate from unit	CT: 0.1016 m/y	Same as recharge rate. Infiltration rate should equal recharge rate (model default).	Infiltration rate estimation can be improved by comparing HELP-model assumptions to actual FFC waste management conditions.
DEPTH, depth of landfill	CT: 3.89 m	Based on median ash generation rate from EPRI Oil Ash Report. See unit area.	
waste fraction	CT: 100 %	Assumes monofill.	
waste density	CT: 1.19 g/cm^3	Assumed.	Same as coal comanagement waste landfill scenario.
TSOURC, duration of leaching	Derived	Model default.	Model default; assumes all contaminants will leach out.
Metals-Specific Variables: Same as Scenario CS (Table A-1)			

Table A-5. EPACMTP Model Inputs: Oil-Fired Utility Waste Onsite Monofill Scenario (Scenario OM) (Continued)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
Unsaturated Zone Variables: Same as Scenario OS (Table A-4)			
Saturated Zone Parameters: Same as Scenario OS (Table A-4)			

Table A-6. EPACMTP Model Inputs: Oil-Fired Utility Waste Offsite Subtitle D Landfill Scenario (Scenario OL)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
Source-Specific Variables			
AREA, management unit area	CT: 34,400 m^2	Same value as non-utility landfill (Scenario NL, Table A-9).	Assumed to represent commercial offsite landfill.
CZERO, leachate concentration	Constituent specific	EPRI Oil Ash Database	Same concentrations used in all oil ash management scenarios.
Cw/Cl value (waste to leachate concentration)	Constituent dependent	EPRI Oil Ash Database	Waste-to-leachate concentrations were calculated for paired sites, although not necessarily from the same sample (i.e., measurements of total and leachate concentrations). A median Cw/Cl value was determined from this array.
RECHRG, recharge rate	CT: 0.1016 m/y	Based on HWIR distribution for silty loam soil. Assigned 1 of 97 climatic centers to each of the 15 generating facilities in the oil ash report (based on its location), and calculated overall statistics. Assumed each generator uses a proximate landfill.	
SINFIL, infiltration rate from unit	CT: 0.1016 m/y	Same as recharge rate. Infiltration rate should equal recharge rate (model default).	Infiltration rate estimation can be improved by comparing HELP-model assumptions to actual FFC waste management conditions.
DEPTH, depth of landfill	CT: 2.25 m	Same as non-utility landfill assumption.	
waste fraction	CT: 24 %	Calculated from central tendency waste quantity from EPRI oil ash report.	

Table A-6. EPACMTP Model Inputs: Oil-Fired Utility Waste Offsite Subtitle D Landfill Scenario (Scenario OL) (Continued)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
waste density	CT: 1.19 g/cm^3	Assumed.	Corresponds to 1 ton=1 cubic yard. Slightly lower than densities in 1988 Report to Congress for dry ash (wet ash is expected to be less dense).
TSOURC, duration of leaching	Derived	Model default.	Model default; assumes all contaminants will leach out.
Metals-Specific Variables: Same as Scenario CS (Table A-1), except as noted below			
USPH, Soil and aquifer pH	CT: 6.80	Median HWIR value. Same as PH variable, below.	
LOM, concentration of dissolved organic carbon in the waste leachate	CT: 49.8 mg/L	Median HWIR value.	
USNOM, unsaturated zone percentage organic matter (should be same as POM)	CT: 0.105 (%)	Mean HWIR value. Same as POM variable, below.	
Unsaturated Zone Variables: Same as Scenario CS (Table A-1), except as noted below			
DSOIL, thickness of unsaturated zone	CT: 4.65 m	Based on HWIR and weighted to surveyed population. Same basis as variable ZB (aquifer thickness).	

Table A-6. EPACMTP Model Inputs: Oil-Fired Utility Waste Offsite Subtitle D Landfill Scenario (Scenario OL) (Continued)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
POM, % organic matter	CT: 0.105 (%)	HWIR mean value.	
Bulk density	CT: 1.65 g/cm^3	HWIR constant value.	
Saturated Zone Parameters: Same as Scenario OS (Table A-4), except as noted below			
PH, ambient groundwater pH	CT: 6.80	HWIR median value	

Table A-7. EPACMTP Model Inputs: FBC Waste Landfill Scenario (Scenario FL)

EPACMTP Data Element	Valua Used in EPACMTP	Data Source	Justification or Concerns
Source-Specific Variables			
AREA, management unit area	CT: 0.155×10^6 m^2 (38 acres) HE: 0.317×10^6 m^2 (77 acres)	CIBO FBC Survey.	Data from 11 facilities.
CZERO, leachate concentration	Constituent specific	CIBO FBC Survey.	
Cw/Cl value (waste to leachate concentration)	Constituent dependent	CIBO FBC Survey.	The median total waste concentration was divided by the median leachate waste concentration for each constituent. This procedure resulted in a median Cw/Cl value.
RECHRG, recharge rate	CT: 0.0903 m/y	Based on HWIR distribution for silty loam soil. Assigned 1 of 97 climatic centers to each of the 14 FBC landfills (based on its location), and calculated overall statistics.	The HWIR distributions are used because they are available and representative of the United States, and no other source directly provides this information.
SINFIL, infiltration rate from unit	CT: 0.0903 m/y	Same as recharge rate. Infiltration rate should equal recharge rate (model default).	Infiltration rate estimation can be improved (if sensitivity analysis justifies additional effort) by comparing HELP-model assumptions to actual FFC waste management conditions.
DEPTH, depth of landfill	CT: 15.8 m HE: 22.9 m	CIBO FBC Survey.	Based on data from 10 facilities.
waste fraction	Constant: 1	Assumed.	
waste density	CT: 1.19 g/cm^3	Assumed.	Same as coal co-management landfill scenario.

Table A-7. EPACMTP Model Inputs: FBC Waste Landfill Scenario (Scenario FL) (Continued)

EPACMTP Data Element	Valua Used in EPACMTP	Data Source	Justification or Concerns
TSOURC, duration of leaching	Derived	Model default.	Model default; assumes all contaminants will leach out.
Metals-Specific Variables: Same as Scenario CS (Table A-1)			
Unsaturated Zone Variables: Same as Scenario CS (Table A-1)			
Saturated Zone Parameters: Same as Scenario CS (Table A-1), except as noted below			
ZB, aquifer saturated thickness	CT: 7.62 m	Based on HWIR and weighted to surveyed population. Each of the 14 relevant landfills in the CIBO FBC survey was assigned to one of 13 hydrogeologic zones based on the predominant subsurface conditions of the meteorological zone.	
XKX, longitudinal hydraulic conductivity, K_x	CT: 631 m/y	Based on HWIR and weighted to surveyed population. Same basis as variable ZB (aquifer thickness).	
GRADNT, hydraulic gradient	CT: 0.005	Based on HWIR and weighted to surveyed population. Same basis as variable ZB (aquifer thickness).	
TEMP, temperature of ambient aquifer water	CT: 15 °C	Based on HWIR and weighted to surveyed population. Same basis as variable ZB (aquifer thickness).	

Table A-8. EPACMTP Model Inputs: Non-Utility Combustion Waste Onsite Monofill Scenario (Scenario NM)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
Source-Specific Variables			
AREA, management unit area	CT: 7,700 m^2 (1.9 acres) HE: 34,500 m^2 (8.5 acres)	Calculated from median waste quantity from US 90. Dimensions are based on waste volume and well-designed landfill. A high-end area, together with high-end depth, represents the overall high-end case.	
CZERO, leachate concentration	Constituent specific.	EPRI site investigations. Same concentrations as used for coal co-management scenarios.	
Cw/Cl value (waste to leachate concentration)	Constituent dependent	EPRI site investigations. Same Cw/Cl values as used for coal comanagement scenarios.	
RECHRG, recharge rate	CT: 0.1143 m/y HE: 0.0005 or 0.4384 m/y	HWIR distribution for silty loam soil: 10th, 50th, and 95th percentile values. Inherent assumption of uniform distribution of FFC sites across 97 climatic centers.	The HWIR distributions are used because they are available and representative of the United States, and no other source directly provides this information. The distributions can be made more specific to FFC sites by weighting the distribution according to the facility locations found from the comanagement survey. This will be pursued if recharge rate is found to have a significant effect on results. Finally, the 10th percentile is used here because the 5th percentile is <0.0001 m/y.
SINFIL, infiltration rate from unit	CT: 0.1143 m/y HE: 0.0005 or 0.4384 m/y	Same as recharge rate. Infiltration rate should equal recharge rate (model default).	

Table A-8. EPACMTP Model Inputs: Non-Utility Combustion Waste Onsite Monofill Scenario (Scenario NM) (Continued)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
DEPTH, depth of landfill	CT: 5.3 m HE: 13.3 m	Same basis as landfill area.	
waste fraction	Constant: 1	Assumed.	
waste density	CT: 1.19 g/cm^3	Assumed.	Same as coal comanagement landfill scenario.
TSOURC, duration of leaching	Derived	Model default.	Model default; assumes all contaminants will leach out.
Metals-Specific Variables: Same as Scenario CS (Table A-1), except as noted below			
USPH, Soil and aquifer pH	CT: 6.80	Median HWIR value. Same as PH variable, below.	
LOM, concentration of dissolved organic carbon in the waste leachate	CT: 49.8 mg/L	Median HWIR value.	
USNOM, unsaturated zone percentage organic matter (should be same as POM)	CT: 0.105 (%)	Mean HWIR value. Same as POM variable, below.	
Unsaturated Zone Variables: Same as Scenario CS (Table A-1), except as noted below			
DSOIL, thickness of unsaturated zone	CT: 3.55 m	HWIR: 50th percentile value.	

Table A-8. EPACMTP Model Inputs: Non-Utility Combustion Waste Onsite Monofill Scenario (Scenario NM) (Continued)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
POM, % organic matter	CT: 0.105 (%)	HWIR mean value.	
Bulk density	CT: 1.65 g/cm^3	HWIR constant value.	
Saturated Zone Parameters: Same as Scenario CS (Table A-1), except as noted below			
ZB, aquifer saturated thickness	CT: 7.09 m	HWIR: 50th percentile value of all hydrogeologic conditions.	
XKX, longitudinal hydraulic conductivity, K_x	CT: 473 m/y	HWIR: 50th percentile value of all hydrogeologic conditions.	
GRADNT, hydraulic gradient	CT: 0.005	HWIR: 50th percentile value of all hydrogeologic conditions.	
TEMP, temperature of ambient aquifer water	CT: 12.5 °C	HWIR: 50th percentile value of all hydrogeologic conditions.	
PH, ambient groundwater pH	CT: 6.80	HWIR median value	

Table A-9. EPACMTP Model Inputs: Non-Utility Combustion Waste Offsite Subtitle D Landfill Scenario (Scenario NL)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
Source-Specific Variables			
AREA, management unit area	CT: 34,400 m^2	Based on subset of Industrial D data base used for HWIR.	Limited statistics to landfills at facilities with SIC codes 26, 28, 49, 20, 82, 33, 22, 37. These are the industries burning coal in the largest quantities, ranked highest to lowest.
CZERO, leachate concentration	Constituent specific.	EPRI site investigations. Same concentrations as used for coal co-management scenarios.	
CZERO, leachate concentration	Constituent specific.	EPRI site investigations. Same concentrations as used for coal co-management scenarios.	
RECHRG, recharge rate	CT: 0.1143 m/y HE: 0.0005 or 0.4384 m/y	HWIR distribution for silty loam soil: 10th, 50th, and 95th percentile values. Same as non-utility monofill scenario.	
SINFIL, infiltration rate from unit	CT: 0.1143 m/y HE: 0.0005 or 0.4384 m/y	Same as recharge rate. Infiltration rate should equal recharge rate (model default).	
DEPTH, depth of landfill	CT: 2.25 m	Estimated from Industrial D facility population.	
waste fraction	CT: 0.56	Calculated from median US90 waste quantity. Same waste quantity as used for non-utility monofill.	
waste density	CT: 1.19 g/cm^3	Assumed.	Same as coal combustion comanagement waste landfill scenario.

Table A-9. EPACMTP Model Inputs: Non-Utility Combustion Waste Offsite Subtitle D Landfill Scenario (Scenario NL) (Continued)

EPACMTP Data Element	Value Used in EPACMTP	Data Source	Justification or Concerns
TSOURC, duration of leaching	Derived	Model default.	Model default; assumes all contaminants will leach out.
Metals-Specific Variables: Same as Scenario NM (Table A-8)			
Unsaturated Zone Variables: Same as Scenario NM (Table A-8)			
Saturated Zone Parameters: Same as Scenario NM (Table A-8)			

APPENDIX B

CALCULATION OF HBLs

APPENDIX B. CALCULATION OF HBLs

Table B-1. HBLs and MCLs for the FFC2 Screening Assessment

Constituent	RfD[a] (mg/kg/d)	Carcinogen Slope factor[a] $(mg/kg/day)^{-1}$	Health-Based Number (mg/L)		MCL[b] or Action Level (mg/L)
			RfD-Based	CSF-Based	
Antimony	0.0004	–	***0.021***	–	0.006 (1° MCL)
Arsenic	0.0003	1.5	0.015	***0.00029***	0.05 (1° MCL)
Barium	0.07	–	***3.60***	–	2 (1° MCL)
Beryllium	0.005	4.3	0.26	***0.0001***	0.004 (1° MCL)
Cadmium	0.0005	–	***0.026***	–	0.005 (1° MCL)
Chromium VI	0.005	–	***0.26***	–	0.1 (1° MCL)
Copper	–	–	–	–	***1.3*** (action level)
Lead	–	–	–	–	***0.015*** (action level)
Mercury	0.0003	–	***0.015***	–	0.002 (1° MCL)
Nickel	0.02	–	***1.03***	–	–[d]
Selenium	0.005	–	***0.257***	–	0.05 (1° MCL)
Silver	0.005	–	***0.257***	–	–
Thallium	0.00008	–	***0.0041***	–	0.002 (1° MCL)
Vanadium[c]	0.007	–	***0.360***	–	–
Zinc	0.3	–	***15.4***	–	–

a. Sources of RfDs and CSFs is IRIS (June 1997) unless otherwise noted.
b. Sources of primary MCLs, secondary MCLs, and action levels are as follows:
   1° MCL for arsenic: 40 CFR 141.13
   All other 1° MCLs for inorganics: 40 CFR 141.62
   Action levels for arsenic and lead: 40 CFR 141.80
c. RfD for vanadium is from HEAST.
d. MCL for nickel has been vacated (60 FR 33932, June 29, 1995) and is therefore not presented in this table.
Bold and *italicized* values are used in all subsequent ground water risk analyses.

For carcinogens, the health-based number (adult) is calculated from the following equation:

$$HBN = \{risk \times BW \times AT \times 365\}/\{I \times ED \times EF \times CSF\},$$

or $$HBN = 4.4 \times 10^{-4}/CSF,$$

where

$risk = 10^{-6}$

adult body weight (BW) = 72 kg (mean value of male and female adults, EPA 1996)
averaging time (AT) = 75 years (average life expectancy for males and females, ibid.)
drinking water consumption rate (I) =1.4 L/d (mean tap water ingestion, ibid.)
exposure duration (ED) =9 year (50th percentile of residence time, ibid).
exposure frequency (EF) = 350 days/year (inherent assumption of assessing adult resident).

For noncarcinogens (i.e., all constituents with RfDs), the health-based number is calculated from the following equation:

$$HBN = \{HQ \times BW \times RfD\}/I,$$

or $$HBN = 51.4 \times RfD$$

where

hazard quotient (HQ) = 1
body weight (BW) = 72 kg for adult (as above)
drinking water consumption rate (I) = 1.4 L/day for adult (as above)

APPENDIX C

SCREENING ANALYSIS

APPENDIX C. SCREENING ANALYSIS

Coal-fired Utility Comanaged Wastes

Table C-1. Screening Results for Coal-fired Utility Comanaged Wastes, EPACMTP and non-EPACMTP Constituents

Constituent	HBL (mg/l)	95th Percentile Observed Concentration (mg/l)	Screening Result and Conclusion
Antimony	0.021	Not detected	0
Arsenic	0.00029-(c)	9.64	***33,241***
Barium	3.6	27.4	***7.61***
Beryllium	0.0001-(c)	Insufficient data	--
Boron	4.63	342	***73.9***
Cadmium	0.026	0.156	***6.00***
Chromium III/VI	0.26	0.746	***2.87***
Copper	1.3-(a)	0.69	0.53
Fluoride	3.08	410	***133***
Lead	0.015-(a)	0.468	***31.2***
Manganese	7.2	103	***14.3***
Mercury	0.015	0.000796	0.053
Molybdenum	0.257	11.4	***44.4***
Nickel	1.03	8.33	***8.09***
Nitrate (as N), total	10-(1)	1170	***117***
Nitrite (as N), total	10-(1)	461	***46.1***
Selenium	0.257	1.03	***4.01***
Silver	0.257	Not detected	0
Strontium	30.8	16.1	0.52
Thallium	0.0041	Insufficient data	--
Vanadium	0.36	0.8	***2.22***
Zinc	15.4	23.1	***1.50***

Table Notes: Numbers in bold and italics indicate which constituents exceed the screening criteria.

All HBL's listed above are non-carcinogens, except for arsenic and beryllium -(c); lead and copper are action levels -(a), not health based numbers; all MCL's are followed by a -(1) or a -(2), indicating primary and secondary, respectively.

Oil-fired Utility Comanaged Wastes

Table C-2. Screening Results for Oil-fired Utility Comanaged Wastes, EPACMTP and non-EPACMTP Constituents

Constituent	HBL (mg/l)	95th Percentile Observed Concentration (mg/l)	Screening Result and Conclusion
Antimony	0.021	Insufficient data	--
Arsenic	0.00029-(c)	4.15	***14,310***
Barium	3.6	12.9	***3.58***
Beryllium	0.0001-(c)	Insufficient data	--
Boron	4.63	Insufficient data	--
Cadmium	0.026	0.62	***23.9***
Chromium III/IV	0.26	3.44	***13.2***
Copper	1.3-(a)	3.42	***2.63***
Fluoride	3.08	0.23	0.075
Lead	0.015-(a)	13.4	***893***
Manganese	7.2	5.16	0.70
Mercury	0.015	0.5	**33.3**
Molybdenum	0.257	Insufficient data	--
Nickel	1.03	470	***456***
Nitrate (as N), total	10-(1)	0.2	0.020
Nitrite (as N), total	10-(1)	0.2	0.020
Selenium	0.257	0.37	***1.44***
Silver	0.257	0.15	0.58
Strontium	30.8	Insufficient data	--
Thallium	0.0041	Insufficient data	--
Vanadium	0.36	882	***2,450***
Zinc	15.4	8.12	0.53

Table Notes: Numbers in bold and italics indicate which constituents exceed the screening criteria.

All HBL's listed above are non-carcinogens, except for arsenic -(c); lead and copper are action levels -(a), not health based numbers; all MCL's are followed by a -(1) or a -(2), indicating primary and secondary, respectively.

FBC Utility Comanaged Wastes

Table C-3. Screening Results for FBC Utility Comanaged Wastes, EPACMTP and non-EPACMTP Constituents

Constituent	HBL (mg/l)	95th Percentile Observed Concentration (mg/l)	Screening Result and Conclusion
Antimony	0.021	1.29	***61.4***
Arsenic	0.00029-(c)	0.35	***1,207***
Barium	3.6	2.6	0.72
Beryllium	0.0001-(c)	0.28	***2,800***
Boron	4.63	3.95	0.85
Cadmium	0.026	0.09	***3.46***
Chromium III/VI	0.26	0.29	***1.12***
Copper	1.3-(a)	0.16	0.12
Fluoride	3.08	Insufficient data	--
Lead	0.015-(a)	0.49	***32.7***
Manganese	7.2	10.9	***1.51***
Mercury	0.015	0.01	0.67
Molybdenum	0.257	0.72	***2.80***
Nickel	1.03	0.42	0.41
Nitrate (as N), total	10-(1)	Insufficient data	--
Nitrite (as N), total	10-(1)	Insufficient data	--
Selenium	0.257	0.26	***1.01***
Silver	0.257	0.13	0.51
Strontium	30.8	Insufficient data	--
Thallium	0.0041	0.07	***17.1***
Vanadium	0.36	1.64	***4.56***
Zinc	15.4	4.46	0.29

Table Notes: Numbers in bold and italics indicate which constituents exceed the screening criteria.

All HBL's listed above are non-carcinogens, except for arsenic and beryllium -(c); lead and copper are action levels -(a), not health based numbers; all MCL's are followed by a -(1) or a -(2), indicating primary and secondary, respectively.

APPENDIX D

POPULATION SURROUNDING FFC SITES

APPENDIX D. POPULATION SURROUNDING FFC SITES

MEMORANDUM

DATE: April 20, 1998

TO: Chris Long

FROM: Cary Comer

SUBJECT: Demographic Study of Areas Surrounding Coal- and Oil-Fired Utilities and Coal-Fired Non-Utilities

Under EPA Contract 68-W4-0030, Work Assignment No. 219, QRT No. 9-3, SAIC developed human population statistics for coal-fired utilities and non-utilities, as well as oil-fired utilities. To estimate the number of people living near fossil fuel combustion (FFC) waste management units, we used the Landview II census software and the location information for all coal- and oil-fired plants in the U.S. to calculate the number of people within 1- and 5-mile radii from the plants. Landview also permitted the compilation of a variety of other demographic characteristics including: population demographics, racial makeup and age distribution. This memorandum summarizes the population statistics for three FFC sectors.

Methodology

Using the EEI Power Statistics database (for operating year 1994), SAIC developed a database including the name, state, county, capacity, fuel usage, and location (latitude and longitude) for all facilities identified as primarily coal-fired and primarily oil-fired and active status with fuel consumption greater than zero for the base year (1994). Similarly, SAIC added to the file all coal-fired non-utilities from the National Interim Particulate Inventory (US90) database. In total, the database contained location information for 453 coal-fired utilities, 84 oil-fired utilities, and 842 coal-fired non-utilities.

The Landview II Census software was then utilized to obtain demographic information for areas surrounding each facility at one- and five-mile radii. The data contained within this software were derived from the 1990 U.S. Census. Using the site-specific latitude and longitudes, Landview provided the following information: total population, total households, racial make-up and age distributions.

For this exercise, SAIC examined the number/percentage of facilities from each FFC sector surrounded by a total residential population of less than 1,000 people, 1,000 to 2,000 people, 2,000 to 3,000 people, and so forth. Tables 1 and 2 present the findings for populations living within 1 mile and 5 miles of the identified plants. Figures 1, 2, and 3 represent the same information for each FFC sector; each Figure simplifies comparison of statistics by comparing data from the 1-mile and 5-mile search areas.

In comparing the 1-mile and 5-mile radius statistics, SAIC found that demographics for non-utility and oil-fired utility plants were very similar. Only 1/3 of these facilities were found to be remotely located with less than 1,000 people living within one mile. Further, nearly all non-utilities (over 90 percent) were found to have greater than 5,000 people living within five miles, and over 80 percent of oil-fired utilities were found in areas with over 20,000 people located within five miles. In contrast, nearly seventy percent of coal-fired utilities showed a population of less than 1,000 residents within one mile of the plant. This difference may reflect the larger size of coal-fired utility plant sites (which typically maintain 10 to 90 days worth of fuel on site and have large solid waste management facilities). Likewise, less than forty percent of the coal-fired utility plants were found to be located in areas with greater than 20,000 residents within five miles of the plant.

Table 1. Distribution of Utilities by Population and Population Totals within 1 Mile

Population Interval	Coal-Fired Utilities		Oil-Fired Utilities		Coal-Fired Non-Utilities	
	# utilities	% utilities	# utilities	% utilities	# utilities	% utilities
0 to 1000	311	68%	29	35%	276	33%
1001 to 2000	51	11	6	7	102	12
2001 to 3000	22	5	6	7	73	9
3001 to 4000	15	3	5	6	41	5
4001 to 5000	9	2	7	8	52	6
> 5000	45	10	31	37	299	35
Total	453	100	84	100	843	100
Total Population within 1 Mile	836,097		1,209,877		4,468,898	
Total Households within 1 Mile	316,827		537,821		1,709,904	
Total Children (<20) within 1 Mile	245,400		255,608		1,252,565	

Table 2. Distribution of Utilities by Population and Population Totals within 5 Miles						
Population Interval	**Coal-Fired Utilities**		**Oil-Fired Utilities**		**Coal-Fired Non-Utilities**	
	# utilities	**%utilities**	**# utilities**	**%utilities**	**# utilities**	**%utilities**
0 to 1000	41	9%	3	3%	17	3%
1001 to 2000	27	6	1	1	19	2
2001 to 3000	32	7	2	2	8	1
3001 to 4000	27	6	0	0	16	2
4001 to 5000	18	4	0	0	14	2
> 5000	308	68	78	94	769	91
Total	453	100	84	100	843	100
Total Population within 5 Miles	21,145,342		21,096,450		74,431,550	
Total Households within 5 Miles	8,119,810		8,533,660		29,108,289	
Total Children (<20) within 5 Miles	6,009,157		5,178,728		20,526,470	

Figure 1. Distribution of Coal-Fired Utilities (n=453 sites)

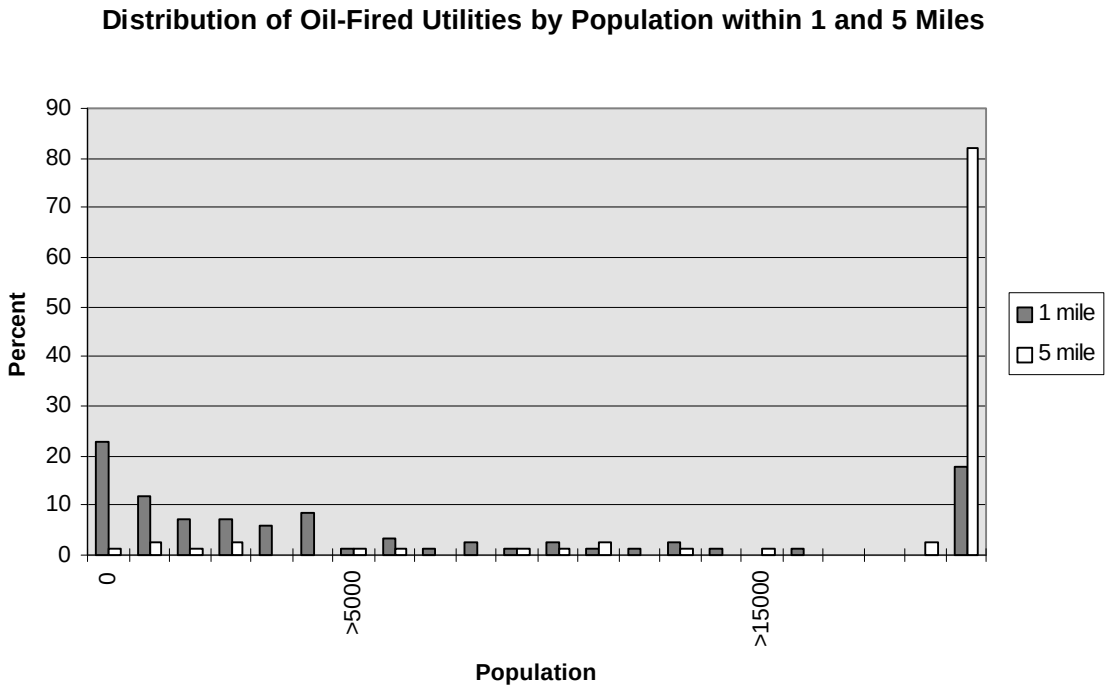


Figure 2. Distribution of Oil-Fired Utilities (n=84 sites)

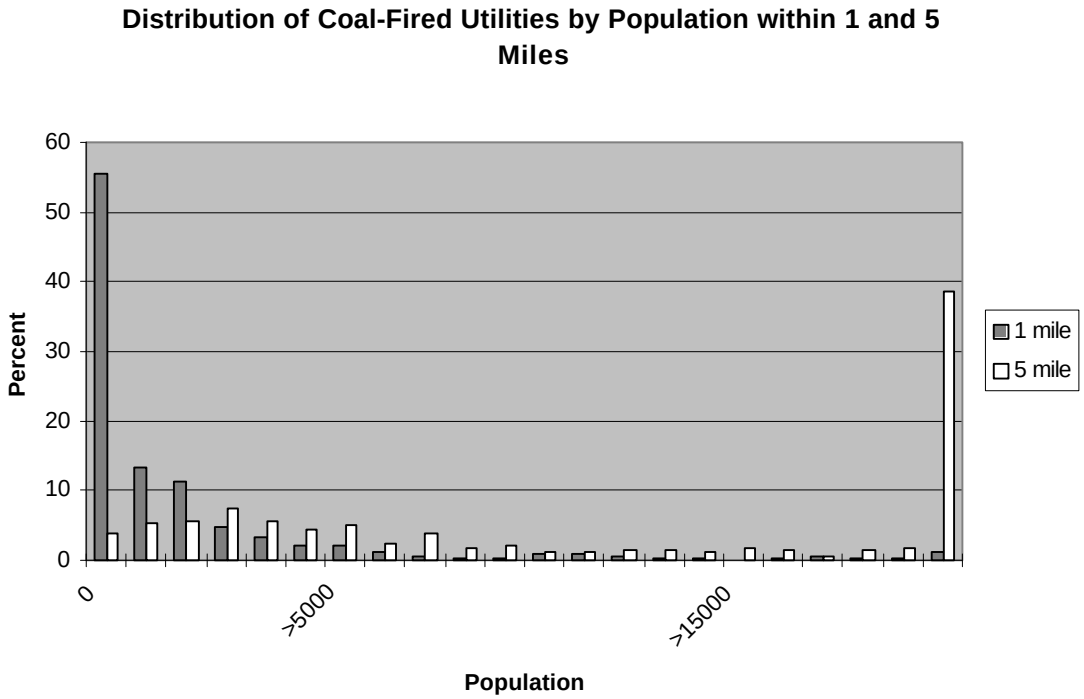
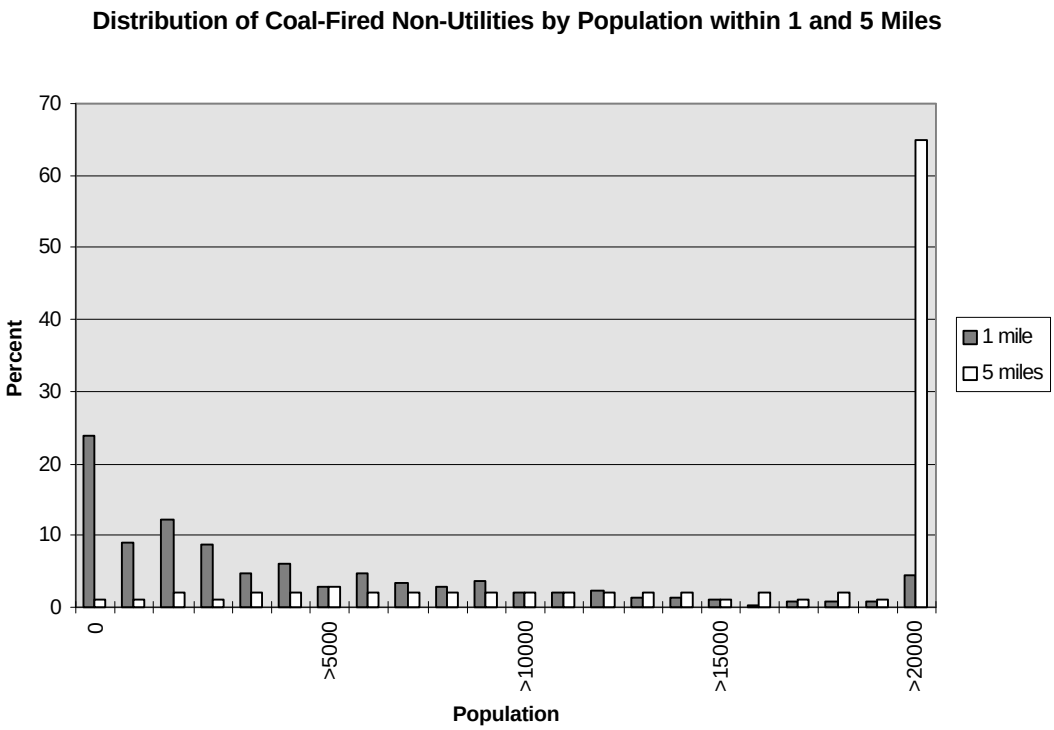


Figure 3. Distribution of Coal-Fired Non-Utilities (n=842 sites)



APPENDIX E

METEOROLOGY AND HYDROLOGY FOR FFC SITES

APPENDIX E. METEOROLOGY AND HYDROLOGY FOR FFC SITES

BACKGROUND

Although many of the variables in EPACMTP describe meteorological and hydrogeological conditions, most of these variables are assumed to be independent of each other. For example, ground water pH is determined and selected independent of the actual site location. In other words, none of the modeling runs in this report assume there is any relation between ground water pH and location. Seven of the variables, however, are related to the actual site location. Therefore, for these seven variables, the actual location of the waste management unit will affect the parameter value. Because the geographic profile of each FFC sector is different (e.g., oil combustion utilities are located in different regions of the country than coal combustion utilities), the values for these seven parameters are also different for each FFC sector and scenario.

DISCUSSION OF SPECIFIC PARAMETERS

All EPACMTP ground water modeling runs for all scenarios and wastes rely on data compiled for HWIR for the following seven parameters referenced above:

- Infiltration rate (all scenarios except surface impoundments)
- Recharge rate
- Depth to ground water (not all EPACMTP runs used HWIR data[1])
- Depth of aquifer
- Aquifer hydraulic gradient
- Hydraulic conductivity of the aquifer
- Ground water temperature.

The first two parameters are assigned values for use in EPACMTP based on the facility's meteorological location (e.g., location in the U.S. where different rainfall is experienced). In the

[1] All deterministic runs for coal co-management wastes, two of the three deterministic runs for oil ash combustion wastes (i.e., the surface impoundment and the onsite monofill scenarios) and the FBC landfill scenario relied on depth to ground water data derived from the 1984 "Environmental Settings" report by EPRI. Subsequent sensitivity analyses showed that the value for depth to ground water has little to no effect on receptor well concentration results for the metal species modeled. Therefore subsequent Monte Carlo runs for these and all other scenarios relied on the HWIR data for consistency in applying the Monte Carlo approach to the remaining six variables.

approach used in this report (identical to the HWIR approach), the U.S. is divided into 97 meteorological locations, each corresponding to a given set of infiltration and recharge rates. The next four parameters are assigned values in EPACMTP based on the facility's hydrogeological location (e.g., the general aquifer characteristics for the particular county). The HWIR approach (used in this report) categorizes the U.S. into 12 general aquifer types, plus a 13th zone representing unknown characteristics. The last parameter (ground water temperature) is also assigned a value based on the facility's geographic location. In summary, two location parameters (for each of meteorological and hydrogeological locations) dictate the values used for the above seven parameters.

METHODOLOGY FOR ASSIGNING METEOROLOGICAL AND HYDROGEOLOGICAL LOCATIONS

For each scenario, a list of waste management unit locations were compiled. The location of these facilities was sometimes limited to the state level (e.g., for coal combustion waste management units) but was known in more detail for other sectors (e.g., oil ash management facilities). However, the methodology for this report used in all cases did not require resolution beyond the state level.

To demonstrate the overall methodology, the following approach describes the steps taken to ascribe a meteorological and hydrogeological classification to each of the 110 landfill and 107 surface impoundment facilities used for coal combustion waste management. A similar approach was used in assigning meteorological and hydrogeological classifications to the other sectors and scenarios.

A list of all facilities (i.e., including both landfills and surface impoundments) was generated and arranged by state. For each state containing at least one management facility, the location number of the meteorological station within that state was determined by referring to a list of the actual locations of the 97 meteorological stations. Meteorological station numbers were then ascribed to each facility in the state. When there were no stations in a state (e.g., Alabama), a meteorological station from a neighboring state was used. If there was only one station in the state (e.g., Iowa), all facilities were assigned this station number. In the majority of cases, there was more than one station per state. When there were fewer facilities than stations in the state (e.g., Arizona and Colorado), station numbers were arbitrarily chosen from the list of stations in the state. When

there were more facilities than stations (e.g., Iowa, Kentucky), station numbers were distributed evenly among the stations.

To determine the hydrogeological location associated with each facility, we referred to a previously compiled list of the 790 landfills used in the Industrial D program where meteorological station and corresponding hydrogeological classification had been assigned to each landfill. We simplified the analysis by assuming a correlation between meteorological location and the overall hydrogeological conditions for that particular part of the country (i.e., an alternative approach would require the research and assignment of aquifer characteristics to each facility). This list of the Industrial D landfills allowed us to determine which hydrogeological locations corresponded to the various meteorological stations assigned to the co-management facility. A hydrogeologic category was then assigned to each co-management facility. In most cases, there was more than one hydrogeological category associated with each meteorological station (i.e. of the four landfills associated with meteorological station 3 in Colorado, three were classified as category 5 and one as category 13). In these cases, the frequency and distribution of categories were used to weigh the assignment of hydrogeological categories to the facilities associated with each station, in a similar manner as described for the meteorological stations. The ground water temperature most frequently associated with the given meteorological location was also extracted from the Industrial D landfill file and assigned to each of the co-management facilities.

USE OF DATA IN DETERMINISTIC AND MONTE CARLO ANALYSES

At the end of the procedure described above, every facility has been assigned a unique hydrogeological and meteorological location. The Monte Carlo analysis used these locations directly in selecting values for the seven parameters described above and therefore no additional calculations were necessary. However, the deterministic analyses required discrete data points for all parameters. The calculation methods used for the co-management deterministic analyses are described below, and are analogous to the approach used for all other scenarios.

The HWIR methodology assigned unique values for infiltration and recharge rate to each meteorological location. Three values are available for each location, depending on the selected soil type. For all deterministic analysis calculations, infiltration and recharge rates associated with silty

loam soil were selected because this is the most predominant soil type in the U.S. according to the EPACMTP Users Guide. Therefore, for each comanagement unit, a single value for infiltration and recharge rates are easily assigned. For the 110 comanagement landfills, the overall median infiltration rate was determined by arraying the infiltration rate values (the recharge rate was assigned a value equal to the infiltration rate as recommended in the EPACMTP Users Guide). Similarly, an overall median ground water temperature was calculated by arraying all ground water temperatures.

The data used as the basis for the remaining four hydrogeological parameters were more detailed and required intermediate calculations. For each of the 13 hydrogeologic locations, median values were determined for each of the four parameters (such as hydraulic gradient), based on the data used in the HWIR approach. These median values were assigned to each of the comanagement units based on the facility's hydrogeological location. Finally, an overall median value for hydraulic gradient and the other three parameters were calculated.

VERIFICATION OF PARAMETER VALUES USED IN THE DETERMINISTIC APPROACH

One verification analysis was conducted to assure that reasonable values for all four hydrogeological parameters were being selected in the deterministic analysis. Specifically, the deterministic approach selected median values for each of four parameters independently, whereas the Monte Carlo approach selects the parameters as measured at a single site. The purpose of the verification analysis was to determine if the set of four values obtained independently could occur at a single site.

The median parameter values used for the deterministic non-utility landfill scenario were compared to the data set used by EPACMTP in the Monte Carlo analysis. The median value for hydraulic conductivity, for example, was determined to be 473 m/y. We extracted all EPACMTP site data with hydraulic conductivity corresponding to 473 m/y, plus or minus 50 percent. For these relatively small number of sites, values for the remaining three parameters (aquifer thickness, depth to ground water, and hydraulic gradient) bounded the calculated median value. Therefore the median values chosen for the deterministic runs are internally consistent with the actual site data. It must be cautioned that such an analysis was not conducted for the high end parameter values, and therefore

a combination of one or more high end values (e.g., high aquifer hydraulic conductivity with thin aquifer) may result in describing an unrealistic site.

APPENDIX F

WASTE CHARACTERIZATION DATA

APPENDIX F. WASTE CHARACTERIZATION DATA

Table F-1 presents an overview of the data sources used to characterize fossil fuel combustion wastes. These data were used as inputs to all scenarios (i.e., landfill, impoundment, and minefill scenarios). The landfill and minefill scenarios require the use of an additional concentration parameter, "Cw/Cl." The Cw/Cl value is defined as the ratio of a metal's concentration in the solid to its concentration in the generated leachate. Cw/Cl represents the total quantity of contaminant available for potential leaching. With all other parameters equal, a higher value of Cw/Cl results in higher receptor well concentrations (and higher risk) than a lower value of Cw/Cl.

Table F-1. Source of Leachate Data for FFC Wastes

Scenario	Data Source
Coal-fired utility comanaged wastes: leachate concentrations for all scenarios	EPRI site investigations (14 sites plus 2 earlier reports) that characterize pore water samples of comanaged wastes within impoundments and landfills
Oil-fired utility wastes: leachate concentrations for all scenarios	TCLP and EP data from EPRI oil ash database supplemented with one verification sample from Florida Power and Light (FP&L)
Fluidized bed combustion (FBC) wastes: leachate concentrations for all scenarios	CIBO data summary tables for FBC byproducts, combined TCLP and EP
Non-utility combustion wastes: leachate concentrations for all scenarios	Same as coal combustion co-management wastes

COAL-FIRED UTILITY COMANAGED WASTE DATA

Leachate characterization data from comanaged coal combustion wastes from the utility industry were compiled from 16 reports, each detailing site investigations from the late 1980s to early 1997. They include the 14 EPRI site investigations, plus 2 additional reports characterizing the comanagement of FGD sludge with low-volume wastes published by EPRI in 1994 (i.e., the "sodium-based FGD sludge" and the "calcium-based FGD sludge" reports). All 16 reports are listed in the bibliography.

These leachate characterization data include pore waters from impoundment wastes and synthetic leachate generated from 2:1 liquid:solid extracts from landfill wastes. Only material sampled

from within the impoundment or within the landfill was considered in this analysis (i.e., samples representing "B" and "C" type samples, representing the full depth of the disposal unit).

Of the 18 sites, 3 represented landfills. Because of this small sample size, EPA did not segregate data representing landfilled wastes from data representing impoundment wastes. For each constituent, the available data at each site were averaged. Each of these site-averaged data points (for up to 18 sites) was arrayed to obtain the desired statistics. Constituents reported as not detected were assigned a value equal to one-half the detection limit. Concentrations for comanaged coal combustion wastes in impoundments and landfills are presented in Table F-2.

Values for Cw/Cl were calculated on a sample-specific basis. EPRI analyzed a set of comanaged waste samples for both total and pore water composition. The ratio of the solid concentration to the pore water concentration was calculated for each sample, for each constituent. Constituents that were not detected were assigned a value equal to one-half the detection limit, but the sample was not used for a given constituent if both the solid and the pore water concentration were reported as not detected. The calculated values of Cw/Cl were arrayed and a median value was determined. (These values are also provided in Table F-2.) In cases where insufficient data were available to calculate Cw/Cl for a specific constituent, a default value of 10,000 was used.

OIL-FIRED UTILITY WASTE DATA

Analytical results on the composition of oil combustion wastes have been provided in EPRI's oil ash database. These data were not supplemented with EPA verification sampling data from FP&L because no TCLP/EP analyses were conducted. Leachate data are available from three different laboratory leach tests: EP, TCLP, and Other. Many constituents were not detected in one or more analyses; in such cases, all measurements identified as below detection limits are assigned concentrations equal to one-half the detection limit. Constituents are presented if they were reported above detection limits in at least one waste sample.

Table F-2. Pore Water Characterization Data for Utility Coal Combustion Wastes in Landfills and Surface Impoundments: Facility-Averaged

Constituent	50th Percentile Observed Conc. (mg/l)	95th Percentile Observed Conc. (mg/l)	Total Number of Samples	Samples below Detection Limit	Number of Sites with Data	Facility with 95th Percentile Concentration	50th Percentile Cw/Cl Value
Constituents Modeled for Ground Water							
Antimony	(f.n. 2)	(f.n. 2)	11	11	2	(f.n. 2)	—
Arsenic	0.0973	9.64	189	52	17	MO	1,200
Barium	0.136	1.04	194	21	17	PA	14,000
Beryllium	(f.n. 3)	(f.n. 3)	11	10	2	(f.n. 3)	—
Cadmium	0.00448	0.156	188	145	17	MO (f.n. 1)	—
Chromium	0.0457	0.746	192	121	18	MO	58,000
Copper	0.037	0.690	148	87	16	MO	13,000
Nickel	0.0883	8.33	193	82	18	MO	17,000
Lead	0.0138	0.468	174	141	15	MO	7,500
Mercury	0.000796	0.000796	4	0	1	SX	—
Selenium	0.121	1.03	176	59	16	Ca FGD report, TT site	42
Silver	(f.n. 2)	(f.n. 2)	95	95	10	(f.n. 2)	—
Vanadium	0.157	0.800	191	37	17	Na FGD report	480
Zinc	0.0825	23.1	203	124	18	MO	3,000
Constituents Not Modeled							
Aluminum	3.87	270	—	—	17	MO	—
Boron	8.02	342	—	—	18	Na FGD report	—
Chloride	62.3	31,600	—	—	18	Na FGD report	—
Fluoride	1.33	410	—	—	17	Na FGD report	—
Iron	0.326	8,100	—	—	17	MO	—
Manganese	0.160	103	—	—	16	MO	—
Molybdenum	0.535	11.4	—	—	18	SX	—
Nitrite (as N), total	0.500	461	—	—	17	O	—
Nitrate (as N), total	2.31	1,170	—	—	15	Na FGD report	—
Strontium	4.46	16.1	—	—	17	Ca FGD report, TT site	—
Sulfate	1,410	115,000	—	—	17	Na FGD report	—

Concentrations at each FFC comanagement site were averaged and the resulting averages arrayed to obtain the median and high-end concentrations presented in this table.

Cw/Cl values are displayed as relevant. A dash (—) indicates that data were not available for calculations or none were needed because no landfill modeling was conducted. A default Cw/Cl value of 10,000 was used in all landfill scenarios where chemical specific values could not be calculated.

f.n. 1: A single non-detect data point from the Na FGD report corresponded to the 95th percentile cadmium concentration for all sites. This data point was deleted from the analysis (to avoid data bias). The 95th percentile concentration of the new data set was determined and presented in this table.

f.n. 2: Concentrations for antimony and silver were not detected at any site. Therefore, concentration data for these two constituents are not presented in this table.

f.n. 3: Beryllium was sampled at only two sites and detected in only one sample. This single data point is likely an outlier. Therefore, concentration data are not presented in this table.

Table F-3 lists the types of TCLP and EP data that are available.

Table F-3. TCLP and EP Data

Waste Type	TCLP		EP	
	# Samples	# Sites	# Samples	# Sites
Bottom Ash	26	12	7	6
Fly Ash	24	9	11	6
Settling Basin Solids	26	15	59	25
Wash Solids [a]	2	2	11	8
Ash, "other"	4	1	---	---
Composite ash	4	3	3	2

a. Includes duct ash, stack ash, air preheater sludge, and air heater wash sludge.

Concentration data are presented in Table F-4. The source of data used for Table F-4 are the summary tables found in Table A-4 of the December 1, 1997, report. Table F-4 represents a compilation of different waste types and leachate procedures. Specifically, separate statistics were generated for each waste type (e.g., median and 95th percentile values for fly ash were determined). A single 95th percentile value was then calculated for each constituent, which reflects the highest 95th percentile TCLP or EP concentration reported for bottom ash, fly ash, settling basin solids, and wash solids. Eight different data sets are combined into this table. A 50th percentile value was calculated for each constituent in the same way (i.e., the highest 50th percentile TCLP or EP concentration reported for bottom ash, fly ash, settling basin solids, and wash solids). This approach overestimates a true median, because waste types with small numbers of samples have similar or identical median and maximum concentrations.

Table F-4. Oil Combustion Wastes - Landfill and Surface Impoundment TCLP/EP Data

Constituent	95th Percentile Observed Concentration (mg/L)	50th Percentile Observed Concentration (mg/L)	Total Number of Samples	Samples below Detection Limit	Number of Sites with Data	50th Percentile Cw/Cl Value
Arsenic	4.15	0.154	171	68	42	830
Barium	12.9	0.49	164	63	41	590
Cadmium	0.62	0.085	173	61	42	100
Chromium	3.44	0.3	179	63	42	750
Manganese	5.16	1.29	---	---	---	---
Mercury	0.50	0.001	168	134	41	---
Nickel	470	470	57	3	23	89
Lead	13.4	0.144	179	83	42	2,200
Phosphorus	2.1	2.1	---	---	---	---
Selenium	0.37	0.0765	173	101	41	3,000
Silver	0.15	0.032	166	102	41	---
Vanadium	882	273	81	7	25	230
Benzene	0.1	0.0038	---	---	---	---
Chloroform	0.025	0.001	---	---	---	---
1,2-Dichloroethane	0.005	0.001	---	---	---	---
Chloride	475	29.4	---	---	---	---
Copper	3.415	0.43	44	5	15	---
Cyanide	0.26	0.26	---	---	---	---
Iron	27.4	1.8	---	---	---	---
Sulfate	1011	408	---	---	---	---
Fluoride	0.23	0.23	---	---	---	---
Zinc	8.12	2.35	50	2	20	---
Methyl ethyl ketone	0.05	0.019	---	---	---	---
Nitrate/nitrite	0.2	0.2	---	---	---	---

Cw/Cl values are displayed as relevant. A dash (---) indicates that data were not available for calculations or none were needed because no landfill modeling was conducted. A default Cw/Cl value of 10,000 was used in all landfill scenarios where chemical specific values could not be calculated.

"Other" ash and composite ash were not included in the calculations. The reasons for omitting these two wastes are as follows:

- **Other ash:** Four "ash-other" waste sampling results were provided in the Oil Combustion By-Products Database. Samples were tested for arsenic, barium, cadmium, chromium, lead, mercury, selenium, and silver. Only barium was detected. The maximum and mean barium concentration levels are below levels detected in bottom and fly ash, and SSB sediments. Therefore, bottom ash, fly ash, and SSB sediments analyzed separately capture the potential risks of "ash-other."

- **Composite ash**: Seven samples of this waste type were analyzed by TCLP and EP. Samples analyzed by EP were below the 95th percentile levels of bottom ash, fly ash, and settling basin solid samples for all analytes. Likewise, samples analyzed by TCLP were below the 95th percentile levels of bottom ash, fly ash, and settling basin solid samples for all analytes except arsenic and methyl ethyl ketone. Omission of these composite ash samples is due to the small number of samples of composite ash and the availability of many other ash samples that should represent concentrations in various combinations of ash.

Values of Cw/Cl were calculated on a sample specific basis. EPRI presented characterization data for total concentrations and TCLP or EP results. Some samples were analyzed for both total and leachate analyses. For these samples, the ratio of the solid concentration to the pore water concentration was calculated for each sample, for each constituent, in the same way as conducted for the coal comanagement data. Constituents that were not detected were assigned a value equal to one-half the detection limit, but the sample was not used for a given constituent if both the solid and the pore water concentration were reported as not detected. The calculated values of Cw/Cl were arrayed and a median value was determined. These values are also provided in Table F-4. In cases where insufficient data were available to calculate Cw/Cl for a specific constituent, a default value of 10,000 was used.

FBC WASTE DATA

Concentrations in Table F-5 are the highest of the following three values presented in the CIBO report: 95th percentile combined ash, 95th percentile bed ash, 95th percentile fly ash. Therefore, the data in this table represent a compilation of multiple data sources. All concentrations are TCLP/EP. Constituents reported as not detected were assigned a value by CIBO equal to one-half the detection limit.

50th percentile concentrations for FBC wastes are calculated in a similar manner to the 95th percentile values for FBC wastes: separate statistics were generated from TCLP/EP data for each of three wastes (combined ash, bed ash, fly ash) and the highest 50th percentile value presented in this table.

Table F-5. FBC Wastes—Landfill TCLP/EP Data

Constituent	95th Percentile Observed Concentration (mg/L)	50th Percentile Observed Concentration (mg/L)	Total Number of Samples	Samples below Detection Limit	Number of Sites with Data	50th Percentile Cw/Cl Value
Arsenic	0.35	0.05	88	39	27	280
Barium	2.6	0.25	94	20	30	2,000
Boron	3.95	0.43	48	8	19	420
Beryllium	0.28	0.025	13	7	7	43
Cadmium	0.09	0.025	76	50	27	160
Chromium	0.29	0.039	90	40	27	480
Manganese	10.9	0.23	53	17	18	5,300
Mercury	0.01	0.001	70	52	26	460
Molybdenum	0.72	0.2	55	10	20	97
Lead	0.49	0.05	82	53	27	160
Antimony	1.29	0.34	48	25	17	1,400
Selenium	0.26	0.05	89	40	30	170
Thallium	0.07	0.05	10	7	5	710
Vanadium	1.64	0.34	15	3	6	920
Aluminum	72.5	2	48	9	14	2,400
Iron	38.8	0.36	53	15	17	2,100
Nickel	0.42	0.037	60	26	22	—
Copper	0.16	0.07	66	29	21	2,600
Silver	0.13	0.025	74	48	27	—
Zinc	4.46	0.075	64	20	21	89

Cw/Cl values are displayed as relevant. A dash (—) indicates that data were not available for calculations or none were needed because no landfill modeling was conducted. A default Cw/Cl value of 10,000 was used in all landfill scenarios where chemical specific values could not be calculated.

Values of Cw/Cl were calculated for each constituent. Unlike the approach used for the oil and coal comanagement combustion wastes, the approach for this scenario did not use individual sample-specific data. Instead, the summary statistics developed by CIBO were employed. For each constituent, the 95th percentile leachate concentration was divided by the 95th percentile solid concentration. The resulting Cw/Cl value represents a central tendency value. These calculated Cw/Cl values are provided in Table F-5. In cases where insufficient data were available to calculate Cw/Cl for a specific constituent, a default value of 10,000 was used.

NON-UTILITY COAL COMBUSTION WASTE DATA

The same data used for utility coal combustion comanaged wastes are used here.

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APPENDIX G

CALCULATION OF WASTE MANAGEMENT UNIT SIZES

APPENDIX G. CALCULATION OF WASTE MANAGEMENT UNIT SIZE

Several of the model input parameters used in EPACMTP ground water modeling were characteristic of the management unit's dimensions or the quantity of waste managed. These model input parameters included the following:

- Management unit area (for all scenarios)
- Management unit depth (for landfill and minefill)
- Waste fraction (for landfill and minefill)
- Ponding depth (for surface impoundment)
- Liner thickness (for surface impoundment)
- Unit duration (for surface impoundment).

This appendix provides the methodology used to calculate the parameters for each of the following scenarios:

- Coal-fired utility comanaged waste surface impoundments
- Coal-fired utility comanaged waste landfills
- Coal-fired utility comanaged waste minefills
- Oil-fired utility waste surface impoundments
- Oil-fired utility waste monofills
- Oil-fired utility waste Subtitle D landfills
- Fluidized bed combustion waste landfills
- Fluidized bed combustion waste minefills
- Non-utility FFC waste monofills
- Non-utility FFC waste Subtitle D landfills.

The appendix also describes how the results were used in each of the deterministic and Monte Carlo analyses.

Management unit dimensions for each scenario were based on industry-specific data, such as surveys. Generally, data from multiple sites were considered to develop median statistics for

deterministic runs, while the full array of site data were used in Monte Carlo analyses. As shown above, surface impoundments required more input parameters and assumptions regarding their operation than did landfills.

Because each of the above parameters were selected independently of each other in the deterministic analysis, a possibility exists that the resultant hypothetical management unit would not be descriptive of an actual management unit. To check for this internal consistency and to verify that the waste management units modeled were sized realistically, the dimensions of the hypothetical unit are compared with the dimensions of actual units. As described for each scenario where sufficient data are available, all management units modeled are within the range of those observed.

COAL-FIRED UTILITY COMANAGED WASTE SURFACE IMPOUNDMENTS

All data for coal combustion waste comanagement impoundments used in this analysis were taken from the 1997 EPRI comanagement survey. The survey covered approximately 100 surface impoundments, although not all of them reported complete data from which to calculate the parameters listed below. The following dimensions were incorporated explicitly in deterministic EPACMTP model runs for this scenario:

- A surface area of 364,000 square meters, or 90 acres, corresponding to the 50th percentile value reported in the EPRI comanagement survey.

- A ponding depth (head depth) of 1.8 meters, or 5.9 feet, and a liner thickness (thickness of the accumulated waste layer) of 3.4 meters, or 11.2 feet, corresponding to the 50th percentile of calculated depths. These depths were not reported in the survey, but were estimated by assuming the impoundment would start out empty and receive an annual quantity of waste equal to that reported on a site-specific basis. The capacity of each impoundment (also reported on a site-specific basis) was used to calculate a total depth, the liner thickness was calculated from the waste generation rate, and ponding depth was determined by difference. To account for changes in depth of accumulated ash over time, a reference year of 20 years was selected as the basis for conducting calculations (corresponding to half of the assumed 40 year impoundment lifetime).

- A source lifetime of 40 years. Throughout the management unit's lifetime the accumulated ash is left in place. The model assumes that all accumulated waste is removed and the impoundment is closed at the end of the 40-year lifetime.

These dimensions used in the deterministic analysis correspond to the following hypothetical impoundment:

- A total depth of 17.1 feet
- An annual waste disposal rate of 81,312 cubic yards
- A total capacity of 2,482,920 cubic yards, with this capacity reached in just over 30 years.

For comparison purposes, all three of these values are just smaller than the median values (40th, 46th, and 42nd percentiles, respectively) calculated for the impoundments in the EPRI comanagement survey. Thus, the hypothetical impoundment is well within the range of dimensions reported in the EPRI population.

Because the EPACMTP assumption of waste removal from impoundments at closure is unrealistic for most larger utility comanaged waste impoundments, the landfill scenario is the more appropriate representation of surface impoundments closed as landfills (i.e., with wastes left in place). As shown for the landfill scenario below, the total landfill capacity exceeds the corresponding surface impoundment capacities. Therefore, the hypothetical landfill may overstate the dimensions of impoundments closed as landfills.

The Monte Carlo procedure used each of the reported surface areas in a separate data file. However, EPACMTP does not allow both liner depth and ponding depth to vary on a site-specific basis; only ponding depth is allowed to vary. Because allowing independent variation of these two related parameters could produce unrealistic physical scenarios, both of these parameters were held constant at the above 50th percentile values through the Monte Carlo runs.

COAL-FIRED UTILITY COMANAGED WASTE LANDFILLS

All data for coal combustion waste comanagement landfills used in this analysis were taken from the 1997 EPRI comanagement survey. The survey covered approximately 100 landfills, although not all of them reported complete data from which to calculate the parameters listed below. The following dimensions were explicitly incorporated in EPACMTP model runs for this scenario:

- A surface area of 267,000 square meters, or 66 acres, corresponding to the 50th percentile value reported in the EPRI comanagement survey.
- An average depth of 9.45 meters, or 31 feet, corresponding to the 50th percentile calculated value. Depth was not reported in the survey, but was calculated from each landfill's capacity and area.
- A waste fraction of 1, indicating that the management unit receives only the comanaged wastes of interest.

These dimensions used in the deterministic analysis correspond to the following hypothetical landfill:

- A total capacity of 3,300,880 cubic yards
- An annual waste disposal rate of 82,522 cubic yards for an assumed 40 year lifetime.

For comparison purposes, the annual waste disposal rate and total capacity are just smaller than the median values (40th and 47th percentiles, respectively) calculated for the landfills in the EPRI comanagement survey. Thus, the hypothetical landfill is well within the range of dimensions reported in the EPRI population.

The Monte Carlo procedure used each of the reported surface areas and average depths in a separate data file, with waste fraction set constant at 1.

COAL-FIRED UTILITY COMANAGED WASTE MINEFILLS

The minefill dimensions assumed for EPACMTP are calculated from 30 minefill projects permitted in the Pottsville Mining District in central Pennsylvania. These data included project area and capacity. Available information did not include ash placement depth. Accordingly, depth was

estimated by dividing project area into project capacity. Original data given in tons were converted to volume assuming a bulk density of 1 ton per cubic yard.

In some cases, no capacity and/or acreage value was reported, and in two cases, the capacity of the minefill was not specified as a volume, but instead as a function of time. These cases were omitted from the calculations of median depth and area. Finally, one site yielded an estimated ash placement depth of 58 meters, or nearly 200 feet. This outlier was omitted from the calculations to avoid excessive influence on the derived median depth.

The median surface area was found to be 141,000 square meters (35 acres). The median depth was found to be 7.56 meters (25 feet). Available data did not permit an estimation of the percentage of total fill material represented by FFC wastes. Therefore, the waste fraction was assumed to be 1 (i.e., the project uses only the comanaged wastes of interest).

No Monte Carlo analyses were conducted for the minefill scenario.

OIL-FIRED UTILITY WASTE SURFACE IMPOUNDMENTS

EPRI's oil combustion report (EPRI, 1998) presents detailed information regarding approximately 15 oil ash management sites, most of them surface impoundments. Based on the review of this information, the following dimensions were incorporated explicitly in deterministic EPACMTP model runs for this scenario:

- A surface area of 3,600 square meters, or 0.90 acres, corresponding to the 50th percentile value for surface impoundment area.
- A ponding depth (head depth) of 1.17 meters, or 3.8 feet, and a liner thickness (thickness of the accumulated waste layer) of 0.21 meters, or 0.68 feet, corresponding to the 50th percentile of calculated depths. As with the coal comanagement impoundment data, these depths were calculated from waste generation, area, capacity, and dredging frequency values reported in the EPRI oil ash report. The oil ash report data shows that surface impoundments for oil combustion wastes are periodically dredged. Therefore, the liner depth was calculated based on the reported dredging frequency and not on a 20-year ash accumulation assumption, as was done for the coal comanaged wastes.

- A source lifetime of 40 years. At the end of the 40-year lifetime the model assumes that all accumulated waste is removed and the impoundment is closed. For oil combustion impoundments, this closure assumption is reasonable.

These dimensions used in the deterministic analysis correspond to the following hypothetical impoundment:

- A total depth of 4.48 feet
- A total capacity of 5,854 cubic yards
- An annual waste disposal rate of 889 cubic yards, applying the median dredging frequency of once per year.

For comparison purposes, the total depth and annual waste disposal rate[1] are smaller than the median values (42nd and 30th percentiles, respectively) calculated for the impoundments in the EPRI oil ash report. The capacity is just larger than the median (55th percentile). Thus, the hypothetical impoundment is well within the range of dimensions reported in the EPRI population.

The Monte Carlo procedure used each of the reported surface areas in a separate data file. Ponding and liner depth were held constant throughout the analysis for the reasons described under coal-fired utility comanaged waste impoundments.

OIL-FIRED UTILITY WASTE MONOFILLS

Because no data directly reported the dimensions of oil combustion waste landfills, the dimensions of the hypothetical landfills were calculated based on the quantity of waste likely to be received. Each facility in the EPRI oil combustion report was assumed to operate its own monofill for the management of settling basin solids, bottom ash, and other oil combustion wastes reported to be generated. The median quantity of waste generated was 825 tons. Corresponding landfill dimensions assumed a density of 1 ton/cubic yard, a constant disposal rates over a 30-year life of the management unit, and a well-designed landfill with a 3:1 side slope and a square pyramid shape with

[1] To compare the derived waste disposal rate to the values reported in the *EPRI Oil Ash Report*, a density of 1 ton to 1 cubic yard was assumed.

a 100-square-foot flat area at the top. As a result, the following dimensions were explicitly incorporated in EPACMTP model runs for this scenario:

- A surface area of 4,860 square meters, or 1.2 acres
- An average depth of 3.89 meters, or 13 feet.

The Monte Carlo analysis used each of the calculated median areas and average depths in a separate data file. Waste fraction was held constant at 1.

OIL-FIRED UTILITY WASTE COMMERCIAL LANDFILLS

A second landfill scenario was modeled for the management of oil combustion wastes, corresponding to the more frequently reported practice of off-site management in a landfill accepting other types of wastes. The following assumptions were used for this scenario:

- The same quantity of waste is disposed in this landfill as in the monofill scenario (i.e., 825 tons for 30 years). Waste fraction is the total volume of waste disposed divided by the total available landfill capacity. The calculated waste fraction is 24 percent.
- The disposal unit is identical to the unit used for the non-utility Subtitle D landfill: an area of 34,400 m^2 and an average depth of 2.25 m.

The Monte Carlo analysis used a constant management unit area and depth but varied waste fraction based on the calculated empirical distribution of reported waste quantity. To retain the Monte Carlo framework regarding meteorological and hydrogeological location used for all other scenarios (as described in Appendix E), each of the oil combustion waste generators was assumed to use a landfill proximate to the point of generation. Therefore, although the Monte Carlo analysis assumed that each landfill has the same area and depth, the location of each landfill was assumed to be different.

FLUIDIZED BED COMBUSTION WASTE LANDFILLS

Data regarding landfill dimensions is available for approximately 15 landfills from the CIBO survey. A few landfills responding to the EPRI comanagement survey reported managing FBC waste. The following dimensions were explicitly incorporated in EPACMTP model runs for this scenario:

- A surface area of 155,000 square meters, or 38 acres, corresponding to the 50th percentile value reported in a combined data set from the CIBO survey and the EPRI comanagement survey.

- An average depth of 15.8 meters, or 52 feet, corresponding to the 50th percentile value derived from capacity and area reported in the combined data set.

These modeled dimensions correspond to the following assumptions about the hypothetical landfill:

- A total capacity of 2,792,747 cubic yards; and
- An annual waste disposal rate of 69,819 cubic yards, assuming a 40 year lifetime.

For comparison purposes, the annual waste disposal rate and total capacity are near the upper end but within the distribution of values reported in the combined data set (66th and 78th percentile, respectively). Thus, the hypothetical landfill is within the range of dimensions reported in the population.

The Monte Carlo analysis used each of these calculated median areas and average depths in a separate data file. Waste fraction was held constant at 1.

FLUIDIZED BED COMBUSTION WASTE MINEFILLS

The same unit dimension data described above for coal comanaged wastes managed in minefills are used for FBC wastes as well. Only a deterministic analysis was conducted.

NON-UTILITY FFC WASTE MONOFILLS

Because no directly reported data on the dimensions of non-utility FFC waste landfills were available, the dimensions of the hypothetical landfills were derived differently from those for other FFC waste management units. Waste generation rates for approximately 850 facilities in the US 90 database were determined based on their reported coal throughput, an assumed 10 percent ash content of coal (i.e., 10 tons of coal produces 1 ton of FFC waste), a density of 1 ton/cubic yard, and a 30 year disposal lifetime. The median 30-year ash generation rate was determined to be 41,200 cubic meters. Using the design assumptions described above for oil-fired utility waste monofills, the

modeled landfill was assumed to have an area of 7,700 square meters and an average depth of 5.3 meters. Waste fraction was 1.

The Monte Carlo analysis used a separate file with calculated areas and depths. Rather than calculate design areas and depths for each facility, however, each landfill was assumed to have a depth of 5.3 meters and the landfill area was calculated from depth and the 30-year waste generation rate. Because the model would not run using exceptionally small areas (represented by generators with very low coal usage rates), the Monte Carlo analysis only considered facilities with calculated areas greater than 3,000 meters. Waste fraction was held constant at 1.

NON-UTILITY FFC WASTE COMMERCIAL LANDFILLS

Landfill dimensions were calculated from a subset of the Industrial D database. The Industrial D data were collected in the 1980s to characterize on-site landfills. To make these data more appropriate for FFC waste management, only the landfills from industries most likely to generate FFC wastes are included. The seven industries considered correspond to SIC codes 20, 22, 26, 28, 33, 37, and 49 (an eighth industry, SIC code 82, was initially considered but no Industrial D landfill data existed for this sector). These seven industries use the largest quantities of coal for non-utility fuel and were therefore assumed to represent on-site FFC waste disposal practices. The 50th percentile median area associated with these facilities is 34,400 square meters. The depth was assumed to be 2.25 meters, which corresponds to the average depth from the Industrial D database for management units of this size ("Potential Risk Due to Air Emissions from Waste Management Units," January 1998, draft, U.S. EPA, page 4-9).

The median waste quantity calculated for the non-utility monofill scenario was used here (i.e, the median 30-year ash generation rate was determined to be 41,200 cubic meters). Using the assumed landfill dimensions described in the previous paragraph, a waste fraction of 0.56 was used in the deterministic analyses.

The Monte Carlo analysis used a constant management unit area and depth but varied waste fraction based on the calculated empirical distribution of reported waste quantity described above for non-utility monofills (waste fraction was capped at 1, in cases where the quantity of waste disposed

exceeded the assumed landfill capacity). To retain the Monte Carlo framework regarding meteorological and hydrogeological location used for all other scenarios (as described in Appendix E), each of the non-utility FFC waste generators was assumed to use an onsite landfill. Therefore, although the Monte Carlo analysis assumed that each landfill has the same area and depth, the location of each landfill was assumed to be different.

APPENDIX H

CHROMIUM SPECIATION AND FFC SITES

APPENDIX H

MEMORANDUM

DATE: February 17, 1998

TO: Dennis Ruddy and Andrew Wittner

FROM: Chris Long and John Vierow

SUBJECT: Draft Review of Chromium Results from FFC Waste EPACMTP Modeling
EPA Contract No. 68-W4-0030, WA No.316
SAIC Project No. 01-0857-08-2060-080

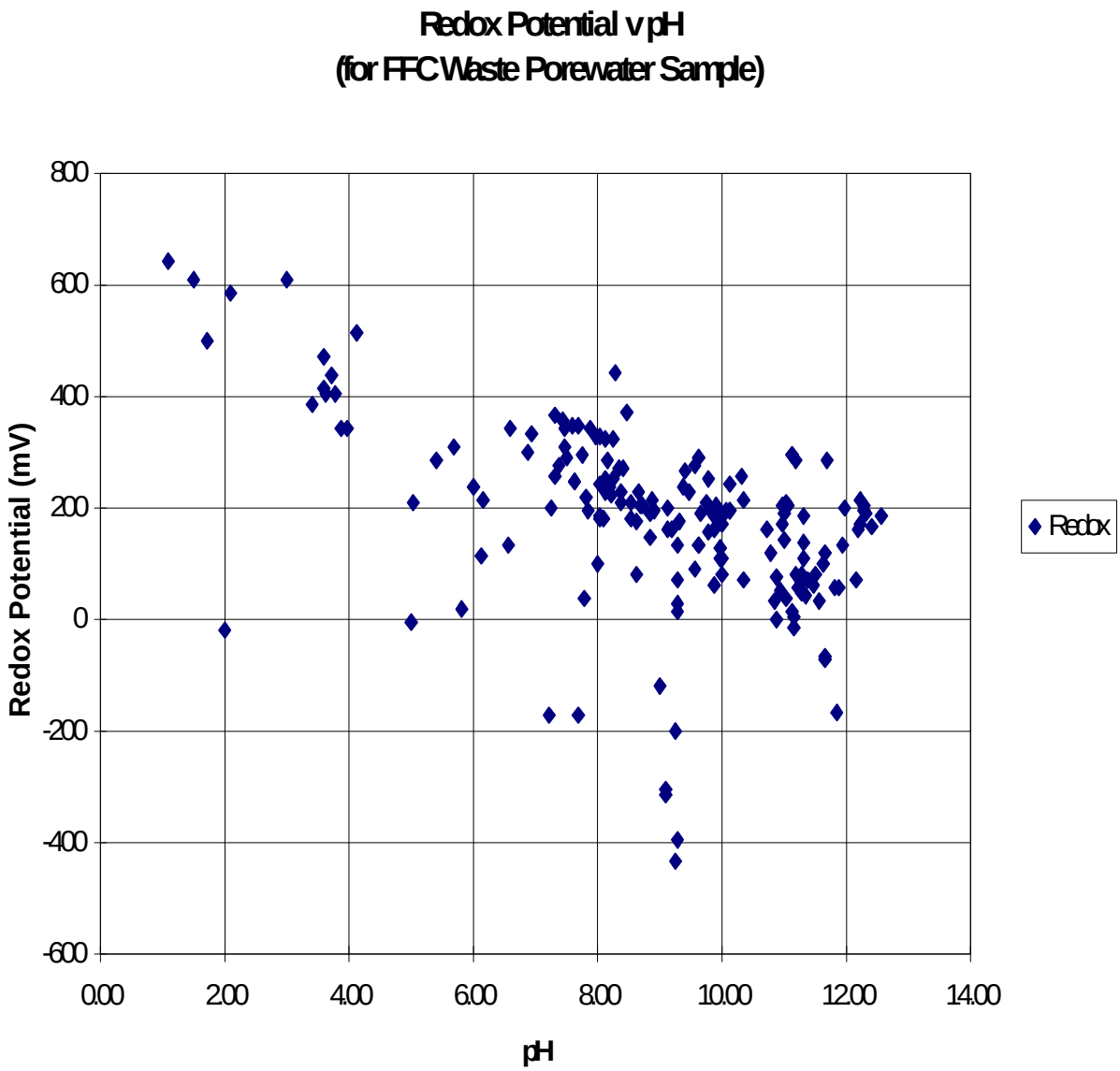
Deterministic scenario modeling results suggest that chromium may migrate from some FFC waste management units. EPACMTP models the two major chromium species, Cr(VI) and Cr(III), as distinct parameters. EPACMTP predicts that the concentration of Cr(VI) may exceed the concentration of the health-based level (HBL) of 0.26 mg/l in groundwater at a point of compliance located 150 meters down gradient of several hypothetical unlined waste management units. However, the model shows virtually no migration potential for Cr(III) in any of the units examined. Because EPA currently lacks detailed chromium speciation data for the environmental samples used to characterize FFC wastes, both the Cr(VI) and Cr(III) predictions result from the same initial input concentrations. Consequently, the current modeling approach necessarily tends to overstate the downgradient concentrations of both chromium species by overestimating the starting concentration of each.

Despite the lack of field observations of speciation, other field observations may be used to suggest the likely valence state of chromium in waste management units. These in turn can be used to qualify the findings of the EPACMTP modeling results.

Various studies describe the influence of pH and oxidation-reduction potential (Eh) on the speciation and solubility of chromium compounds (e.g. Eary, et al 1990; EPRI, 1987; EPRI, 1989; Theis and Gardner, 1990). Cr(VI) is significantly more soluble and mobile than is Cr(III) (EPRI 1989). Generally, however, reducing to moderately oxidizing conditions strongly favor Cr(III) compounds over Cr(VI) compounds (EPRI, 1989), and will readily reduce Cr(VI) to Cr(III). One source indicates that chromium will exist stabily as Cr^{3+}, $CrOH^{2+}$, or $Cr(OH)_3$ when pH is 12 or less and the redox potential is 400 mV or less (Eary, 1990). Another source indicates that Cr(VI) will only be present in significant concentrations when pH is less than 4.5 and pE+pH exceeds 12, or at higher pH when pE + pH is 18 or more (EPRI, 1984).

Figure H-1 shows a plot of pH (in standard units) versus redox potential (in millivolts) for 173 porewater samples collected from 13 comanagement landfills and impoundments. The plot demonstrates that conditions in all of the porewater samples from all of the sites favor formation of Cr(III) compounds over Cr(VI) compounds, based on the above criteria. In fact, field pH measurments rarely exceeded 12 s.u., and the measured redox potential was consistently below 400 mV. It is also worth noting that in more than 100 samples, the measured total chromium concentration was below detection, further indicating the low potential for hexavalent chromium migration at the concentrations predicted in the high-end deterministic modeling. (See Appendix F for more details on chromium concentrations and non-detects.)

Figure H-1. Oxidation-reduction Potential vs. PH



APPENDIX K

EPACMTP MODEL SENSITIVITY

APPENDIX K. EPACMTP MODEL SENSITIVITY

A series of analyses were conducted to determine groundwater risks associated with FFC wastes. This effort included an initial sensitivity analysis of 27 parameters followed by a subsequent double high end parameter analysis for five parameters. Details of the analyses and results are provided below. These results are useful in conjunction with the Monte Carlo analyses presented in Section 5 to evaluate the confidence and uncertainty in the high end evaluations.

EXISTING DATA ON SENSITIVITY ANALYSES

Sensitivity analyses performed for other OSW projects are useful in selecting sensitive parameters for FFC waste analysis. Although these previous results cannot substitute for analyses specific to FFC wastes, they can be used to verify the conclusions of the present analysis. Sensitivity analyses have recently been conducted in conjunction with the following programs:

- Petroleum refining rule. Double high end parameter analyses were conducted to assess ground water risks associated with landfilling of petroleum refining wastes (i.e., values for two input parameters were changed for each run). Results for surface impoundments were not presented. See "Supplemental Background Document for Groundwater Pathway Risk Analysis," March 1997.

- Industrial D program. Ongoing work supporting the development of Industrial Subtitle D guidance has included sensitivity analyses for landfills and surface impoundments, among other units. Specifically, sensitivity analyses were conducted by varying the value for a single parameter and either holding the remaining values constant or running them in Monte Carlo mode.

These two projects have shown the following parameters to be particularly sensitive for scenarios appropriate for metals:

- Disposal unit area
- Receptor well location (both distance from source and angle off centerline)
- Infiltration rate
- Ponding depth (for surface impoundment)
- Waste quantity
- Waste concentration, leachate and total

- Longitudinal hydraulic conductivity of aquifer
- Hydraulic gradient of aquifer
- Aquifer thickness

CURRENT SENSITIVITY ANALYSES

A complete sensitivity analysis would include evaluating multiple combinations of every input parameter, for all chemicals and scenarios. However, significant effort is saved by only considering those input parameters which significantly impact receptor well concentrations. To determine the parameters that significantly affect results, a limited number of runs were conducted for all input parameters. Parameters which do not significantly affect the receptor well concentration in any of these runs are eliminated from further consideration with high confidence. These results were then checked against sensitivity analyses from previous OSW efforts (discussed above) for added confidence.

Sensitivity analyses for all 27 input parameters or sets of parameters were conducted for the following constituents and scenarios[1]:

- Arsenic, beryllium, and nickel in a coal combustion waste co-management surface impoundment. Arsenic represents a linear isotherm constituent, while beryllium and nickel represent non-linear isotherm constituents.
- Arsenic in a coal combustion waste co-management landfill.

Each input parameter was varied, one at a time, between its central tendency value and its high end value. The precise values used as central tendency and high end values are presented later in this Appendix. The result were then compared to the scenario's result when all input parameters were set at central tendency. As expected, different parameters had different sensitivity results depending on the scenario and metal evaluated. Table K-1 summarizes the results of this analysis by presenting the top 10 variables which had the greatest effect for each analysis. The complete results are presented later in this appendix. The following 14 parameters are represented in Table K-1:

[1] For example, initial concentration is used as a single variable in EPACMTP, but the difference between central tendency well location and high end well location were defined by two input variables: distance from the source and its angle off the centerline.

- Initial concentration
- Well location
- Liner conductivity of surface impoundment
- Ponding depth of surface impoundment
- Liner thickness of surface impoundment
- Duration of leaching of surface impoundment
- Acidic aquifer pH
- Hydraulic conductivity of saturated zone
- Aquifer thickness
- Hydraulic gradient of saturated zone
- Recharge rate
- Iron oxide concentration
- Unit area
- Saturated bulk density

These 14 parameters include all of those listed earlier in this appendix when reviewing sensitivity results from previous projects, with the exception of waste quantity (which was not explicitly included in Table K-1). Waste quantity is applicable to a landfill scenario and is used in the model as waste fraction. Waste fraction was held constant in this sensitivity analysis (a value of 100 percent was used).

Table K-1. Sensitivity Analysis Results: Top 10 Variables Affecting Receptor Well Concentration in Coal Co-managed Waste Landfills and Impoundments

Rank (1=most sensitive)	Impoundment			Landfill
	Beryllium	**Arsenic**	**Nickel**	**Arsenic**
1	Well location	Initial concentration	Initial concentration	Initial concentration
2	Initial concentration	Liner conductivity	Liner conductivity	Well location
3	Liner conductivity	Ponding depth	Iron oxide concentration	Saturated bulk density
4	Ponding depth	Liner thickness	Aquifer pH (acidic)	All other parameters have no effect
5	Liner thickness	Duration of leaching	Unit area	
6	Duration of leaching	Saturated hydraulic conductivity	Ponding depth	
7	Aquifer pH (acidic)	Aquifer thickness	Liner thickness	
8	Saturated hydraulic conductivity	Hydraulic gradient	Saturated hydraulic conductivity	
9	Aquifer thickness	Unit area	Duration of leaching	
10	Hydraulic gradient	Recharge rate	Well location	

DOUBLE HIGH END ANALYSES

A series of double high end analyses were conducted by running the surface impoundment scenario with two parameters at their high end values while holding all else at central tendency values. Only five parameters (rather than all variables in Table K-1) were used in this analysis:

- Initial concentration
- Well location
- Liner conductivity of surface impoundment
- Ponding depth of surface impoundment
- Aquifer thickness

These parameters were selected because Table K-1 shows them to be most sensitive for these three metals in this scenario. The results of these runs are summarized in Table K-2. These results show that the combination of liner conductivity and initial concentration result in the highest risk for

all three constituents. However, this analysis used greater differences between the central tendency and high end values for liner conductivity than were used in the other analyses of this report (i.e., a central tendency value of $5x10^{-7}$ cm/s was used in these sensitivity analyses, while a central tendency value of $1x10^{-6}$ cm/s was used in the analyses presented in the rest of this report. In both cases the high end value was $1x10^{-5}$ cm/s. Therefore the high ratios shown for the combination of liner conductivity and concentration may be misleading.

Table K-2. Double High End Analyses Results

Parameters Set at High End	Ratio of Scenario Run to Central Tendency Run		
	Arsenic	Nickel	Beryllium
Concentration / Liner Conductivity	1160	29,290	80
Concentration / Ponding Depth	540	5430	37
Concentration / Well Location	350	3730	23
Concentration / Aquifer Depth	300	3090	20

CONCLUSION

Setting concentration and well location to the high end for surface impoundment scenarios represents a case with significant risk, but not necessarily one with the absolute highest risk. The limited single high end analyses performed for the landfill scenario show concentration and well location to be only two of three parameters affecting the receptor well concentration.

Significant differences between the central tendency and high end initial concentrations (as listed in Appendix F) show initial concentration to be a sensitive parameter. The above results demonstrate the significant impact of leachate concentration on receptor well concentrations and thus demonstrate the significant impact of this parameter on ground water risks associated with coal waste surface impoundments. Additionally, receptor well location is a sensitive parameter for landfill analysis due to the long migration times for these metals (Section 5 shows that the constituents of concern take thousands of years to reach levels of concern for only a 150 meter distance). Therefore, setting well location to a distance greater than 150 meters may result in little to no receptor well risk simply because the constituent has not migrated that far in the 10,000 year EPACMTP evaluation time. However, Chapter 5 also shows that migration time is not as significant for the impoundment scenarios and in fact the analyses presented in this appendix suggest that parameters affecting surface

impoundment infiltration (i.e., liner conductivity and ponding depth) may have greater influence on the receptor risk.

The limited evaluations presented here do not include all sensitivity results. The Monte Carlo results presented in Section 5 are valuable in verifying that the high end analysis actually does represent a high end case. The results presented in this Appendix, on the other hand, are particularly valuable in identifying those parameters that most influence results.

SENSITIVITY ANALYSIS DETAILS

Table K-3 lists all results of the sensitivity analysis. First, a set of central tendency input parameters were assembled and run to obtain a reference result. Then, each parameter was changed, one at a time, between its central tendency and high end result to isolate the effect of this one parameter. Results are expressed as the ratio of the receptor well concentration of the test case relative to the receptor well concentration in the central tendency case. In some cases it was unknown if the 5th percentile value or the 95th percentile value would correspond to the high end case, and therefore the analysis was conducted both ways. Note that a ratio greater than one indicates the test case is more conservative (i.e., actually is a high end analysis), while a ratio less than one means that the test case was less conservative. A ratio equal to one indicates the parameter has no effect on results.

Table K-3. Sensitivity Analyses Results for Surface Impoundment and Landfill Scenarios: Ratio of The Scenario Run to the Central Tendency Run

Parameter Set at High End	Ratio of Scenario Run to Central Tendency Run			
	Surface Impoundments			Landfills
	Beryllium	Arsenic	Nickel	Arsenic
Initial Concentration	12.1	184.5	2065.3	3.5
Well Location [A]	415.9	0.0	1.9	1.5
Liner Conductivity	10.8	6.3	6.3	0.0
Iron Oxide (95%)	1.0	1.0	5.8	1.0
Iron Oxide (5%)	1.0	1.0	1.8	1.0
Ponding Depth	4.5	2.9	2.9	1.0
Liner Thickness	4.4	2.8	2.8	0.0
TSource (100 yr) [B]	3.7	2.5	2.5	NA
TSource (20 yr) [B]	0.1	0.5	0.5	NA
Aquifer pH (5%)	3.5	1.2	3.2	1.0
Aquifer pH (95%)	0.7	0.9	0.0	0.9
Unit Area	1.5	1.4	3.0	1.0
Sat. Conductivity (95%)	2.7	2.4	2.7	0.3
Sat. Conductivity (5%)	0.4	0.4	0.4	0.8
Aquifer Thickness	1.6	1.6	1.6	1.0
Gradient (95%)	1.6	1.5	1.6	1.0
Gradient (5%)	0.8	0.8	0.8	0.9
Depth to Watertable	1.6	0.9	0.9	1.0
Recharge Rate	1.3	1.3	1.3	1.0
Sat. Bulk Density (95%)	1.2	1.2	1.2	1.0
Sat. Bulk Density (5%)	0.9	0.9	0.9	0.9
Unsat.Bulk Density (95%)	1.3	1.0	1.0	1.0
Unsat. Bulk Density (5%)	0.8	1.0	1.0	1.0
Longitudinal Dispersivity	1.1	1.1	1.1	1.0
Porosity (95%)	1.0	1.0	1.0	1.0
Porosity (5%)	1.0	1.0	1.0	1.0
Fraction Org. Carbon (95%)	1.0	1.0	1.0	1.0
Fraction Org. Carbon (5%)	1.0	1.0	1.0	1.0
LOM (95%)	1.0	1.0	1.0	1.0
LOM (5%)	1.0	1.0	1.0	1.0

Parameter Set at High End	Ratio of Scenario Run to Central Tendency Run			
	Surface Impoundments			Landfills
	Beryllium	Arsenic	Nickel	Arsenic
% Organic Matter (5%)	1.0	1.0	1.0	1.0
%Organic Matter (95%)	1.0	1.0	1.0	1.0
Z-Well	1.0	1.0	1.0	1.0
Particle Diameter (5%)	1.0	1.0	1.0	1.0
Particle Diameter (95%)	1.0	1.0	1.0	1.0
Aquifer Temperature (95%)	1.0	1.0	1.0	1.0
Aquifer Temperature (5%)	1.0	1.0	1.0	1.0
Waste Density (low) [C]	NA	NA	NA	1.0
Waste Density (high) [C]	NA	NA	NA	1.0
N value	NA	NA	NA	1.0
Sandy Loam Data [D]	0.9	0.9	0.9	1.0

NA: Parameter not used in scenario.

A Well location central tendency value is assigned a distance of 427 m and a location anywhere in the plume. Well location high end value is assigned a distance of 150 m and an angle fixed at centerline.

B Surface impoundment source duration was arbitrarily varied between 20 and 100 years.

C Waste density was arbitrarily varied between 1.0 and 1.44 g/cm^3.

D Unsaturated zone soil property data for sandy loam were varied as a group.

The central tendency and high end values used in this sensitivity analysis were, in some cases, slightly different than the central tendency and high end values used in the remainder of this report. These differences are not expected to change the conclusion that a particular parameter is sensitive, but may change its importance relative to other parameters. Table K-4 summarizes the differences between central tendency and high end values for parameters included in the sensitivity analyses and the current model runs for both surface impoundments and landfills. The values used in the Chapter 5 model runs are given in Appendix A. The use of different values can be attributed to the fact that the sensitivity analyses were conducted prior to the various model scenario runs. The most significant parameter in Table K-4 is liner conductivity. The central tendency value of liner conductivity changed between these sensitivity analyses and the analyses conducted for Chapter 5, and therefore the influence of this parameter on the results, while still very important, is not known with as much certainty.

Table K-4. Differences between Values Used in Sensitivity Analysis and Values Used for Chapter 5 Results

EPACMTP Data Element	Value Used in Sensitivity Analysis	Value used in Current Report	
		landfills	surface imp.
Initial concentration: arsenic (mg/L)	CT: 0.078 (impoundment) HE: 14.36 (impoundment) CT: 0.0025 (landfill) HE: 0.0087 (landfill)	HE: 9.64	HE: 9.64
Initial concentration: beryllium (mg/L)	CT: 0.001 (impoundment) HE: 0.0062 (impoundment)	HE: 0.0062	HE: 0.0062
Initial concentration: nickel (mg/L)	CT: 0.019 (impoundment) HE: 16.05 (impoundment)	HE: 8.33	HE: 8.33
Cw/Cl value (waste to leachate concentration): arsenic	CT: 1,000 (landfill) HE: 7,000 (landfill)	CT: 1,200	Not applicable
RECHRG, recharge rate	CT: 0.1143 m/y HE: 0.0005 m/y (impoundment)	CT: 0.0894 m/y	CT: 0.3256 m/y
SINFIL, infiltration rate	CT: 0.1143 m/y (landfill) HE: 0.4384 m/y (landfill)	CT: 0.0894	Not applicable
CLINR, hydraulic conductivity of liner	CT: 0.158 m/y HE: 3.16 m/y	Not applicable	CT: 0.315 m/y
waste density	1.26 g/cm^3	CT: 1.19 g/cm^3	Not applicable
ZB, aquifer saturated thickness	CT: 7.09 m HE: 3.66 m	CT: 15.20 m	CT: 15.20 m
XKX, longitudinal hydraulic conductivity, K_x	CT: 473 m/y HE: 31.5 or 11,000 m/y	CT: 315 m/y	CT: 315 m/y
GRADNT, hydraulic gradient	CT: 0.005 HE: 0.002 or 0.02	CT: 0.009	CT: 0.009
TEMP, temperature of ambient aquifer water	CT: 12.5 °C HE: 7.5 or 22.5 °C	CT: 12.5 °C	CT: 17.5 °C

All other values used in sensitivity analysis are same as in Appendix A.

APPENDIX L

DIOXINS AND FURANS IN ASH FROM COAL COMBUSTION

APPENDIX L. DIOXINS AND FURANS IN ASH FROM COAL COMBUSTION

INTRODUCTION

SAIC was asked to review available information regarding the likely occurrence of polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) in coal ash from electric power generating facilities. This review effort occurred in three distinct phases. The first phase, in February 1993, focused on a review of readily available literature, primarily from the journal *Chemosphere*, which is the principal peer-reviewed journal covering dioxin-related issues. The second phase was undertaken in August and September of 1997 and focused on literature published since the 1993 review and investigated a broader base of possible publications. The third phase took place in April 1998, when SAIC reviewed a copy of the Electric Power Research Institute (EPRI) report entitled *"PCDDs and PCDFs in Coal Combustion Byproducts (CCBs),"* dated March 1998. This SAIC report describes the findings of all three phases of the review.

BACKGROUND

PCDDs and PCDFs (often generically referred to as "dioxins" and "furans") are two related classes of compounds of environmental concern. The basic structures of dibenzo-*p*-dioxin and dibenzofuran are shown in Figure L-1. Each of the two aromatic ring structures is made up of six carbon atoms. In the unchlorinated dibenzo-*p*-dioxin, each of these carbon atoms is bonded to two other carbon atoms. Four carbon atoms on each ring are also bonded to hydrogen atoms. In the formal structural nomenclature of these compounds, the carbon atoms are numbered as shown in Figure L-1, where the oxygen atoms occupy the 5 and 10 numbered positions in the structure.

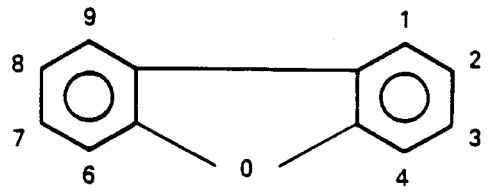


Figure L-1 - Generalized Structures of Dibenzo-*p*-dioxin and Dibenzofuran

Substituting chlorine atoms for one or more of these eight hydrogen atoms results in a chlorodibenzo-*p*-dioxin. When more than one chlorine atom is attached to the basic structure, the

resulting isomer is called a polychlorinated dibenzo-*p*-dioxin. Adding up all the possible PCDDs containing one to eight chlorine atoms, there are a total of 75 possible PCDD isomers. Similar substitutions of chlorine atoms for hydrogen atoms on the dibenzofuran structure shown in the figure result in a total of 135 possible PCDF isomers.

The PCDD isomer with chlorine atoms in the 2,3,7, and 8 positions of its structure is known as 2,3,7,8-tetrachlorodibenzo-*p*-dioxin, or 2,3,7,8-TCDD. This isomer has been shown to cause severe acne in humans, to cause abnormal fetal development (teratogenesis) in mice, to be carcinogenic to rats, and to be acutely toxic to guinea pigs. Media attention in the past 20 years has focused on the characterization of this isomer as "the most toxic organic compound known to man."

Of the total of 210 PCDD/PCDF isomers, the toxicologic concerns center on those 17 PCDD/PCDF isomers that bear chlorine atoms in the 2, 3, 7, and 8 positions of their structures. Within this group of 17 isomers, the toxicologic effects vary widely. In 1987, EPA adopted a procedure for assessing the risks associated with exposures to PCDDs/PCDFs that involved the use of toxic equivalency factors (TEFs) to relate the toxicity of any one of the isomers to that of the most toxic isomer, 2,3,7,8-TCDD. That procedure was updated in 1989, on the basis of an internationally accepted set of TEF values (Barnes *et al.*). The 1989 TEF values range from 1.0 for 2,3,7,8-TCDD (most toxic) to 0.001 for octachlorodibenzo-*p*-dioxin (OCDD, least toxic). Those isomers without chlorine atoms in the 2, 3, 7, and 8 positions are not included in an evaluation of the risk associated with PCDD/PCDF exposure.

In regulating the disposal of wastes under RCRA, EPA has not established concentration-based limits for PCDDs/PCDFs in the identification and listing of a hazardous waste (40 CFR Part 261, July 1, 1990). However, under Sections 260.20 and 260.22, the Agency has granted an exclusion from hazardous waste regulation to rotary kiln ash and other solids that result from the operation of an EPA mobile incinerator at the Denney Farm Site in McDowell, Missouri. This incinerator has been used to treat soils and other materials known to be contaminated with dioxins and furans. This exemption was granted on the provision that the operator could demonstrate that the incinerator ash contained concentrations of PCDDs and PCDFs that, when multiplied by the 1989 TEF values and summed together, were no higher than 5 parts per trillion (ppt) of TCDD.

Initial concern with 2,3,7,8-TCDD arose when this compound was discovered to be a trace contaminant of herbicides such as 2,4-D and of polychlorinated biphenyl (PCB) formulations used in electrical transformers. Found in such products at part per million (ppm, or mg/kg) levels, 2,3,7,8-TCDD was released into the environment during spraying of the herbicide or through disposal of PCBs or PCB-containing equipment. The discovery of part per billion (ppb) concentrations of 2,3,7,8-TCDD in soil samples in Missouri eventually resulted in the purchase and extensive cleanup efforts at Times Beach, Missouri.

In the middle 1970s, the focus of international interest in PCDDs and PCDFs shifted from those inadvertently generated during the industrial production of organic chemicals to concern that these compounds were being identified in an increasingly wide range of environmental samples. This seemingly ubiquitous occurrence of PCDDs and PCDFs in the environment lead to the realization that there must be additional sources of these compounds beyond their presence as trace contaminants of certain organic chemicals and that in order to account for the occurrence of these compounds, these additional sources must be widely distributed.

PCDDs and PCDFs were identified in the by-products and effluents of combustion processes as early as 1977 (Olie *et al*.). Further work by Bumb *et al.* (1980) and others demonstrated that there were important sources of PCDDs and PCDFs beyond the production of certain organic chemicals, and that these sources were responsible for substantial releases of these compounds to the environment. One of those additional sources was the burning of municipal refuse.

Although there has been considerable research aimed at identifying and controlling sources of PCDDs/PCDFs from combustion processes related to incineration, there has been much less attention paid to the combustion of coal or other fuels for power generation.

LITERATURE REVIEW

Phase 1 Findings

The burning of municipal or industrial wastes has been extensively investigated, and the references are too numerous to list here. A variety of PCDDs and PCDFs, including 2,3,7,8-TCDD,

have been identified in various export vectors (flue gases, fly ash, scrubber waters, bottom ash, etc.) from such incineration processes. These results are not unexpected, given that such wastes often contain materials composed of aromatic organics and organochlorine compounds. However, while coal certainly contains aromatic organics, it is not known to contain organochlorine compounds.

The possibility that the combustion of coal for power generation was a PCDD/PCDF source was investigated as early as 1980 (Kimble and Gross), and subsequently by Junk and Richard in 1981. These authors focused on the principal compound of concern at that time, 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (2,3,7,8-TCDD). Neither investigation found any 2,3,7,8-TCDD in the fly ash from the burning of low-sulfur coal or from the burning of a coal-refuse mixture.

Work by Eklund and Stromberg (1983) demonstrated that *some* chlorinated organic compounds could be found in the flue gases from coal-fired boilers, indicating that inorganic chloride in coal could be converted to organochlorine during combustion. These authors concluded that some polychlorinated polynuclear aromatic compounds were produced by the chlorination of their corresponding non-chlorinated analogs. However, they did not detect any PCDDs/PCDFs in their experiments. Their work involved only flue gases. They did not sample either fly ash or bottom ash from these sources.

Chiu *et al.* (1983) studied PCDD/PCDF emissions from a variety of stationary combustion sources, including a large modern coal-fired power generating station and a 30-year old coal-fired heating boiler. They found no PCDDs/PCDFs in the fly ash from the coal-fired heating boiler, and found only low ppb levels (1-32 ppb) of any PCDDs/PCDFs in one sample of fly ash from the power plant. Although the analyses that they conducted were capable of distinguishing the individual PCDD/PCDF isomers, tha data that they present are given as total TCDD, total TCDF, etc. Based on their data, they concluded that there was insufficient evidence to determine if this one power plant was a potential source of PCDDs/PCDFs.

In an attempt to resolve the debate about coal combustion sources of PCDDs/PCDFs, Czuczwa and Hites (1984) examined two types of fly ash from coal-fired power plants. They found no tetrachloro- or pentachlorodioxins or furans in the fly ash samples studied, and only traces of hepta

and octachlorinated isomers (HpCDD, HpCDF, OCDD, and OCDF). The levels of these latter compounds were at least 100 times lower than the reported concentrations in municipal incinerator ash (i.e., 2-4 ppb of OCDD were found in coal fly ash samples, while over 400 ppb of OCDD were found in municipal incinerator ash.) Considering these results in light of the 1989 TEF values, the toxicity of these concentrations of OCDD would be three orders of magnitude lower (i.e., 2-4 parts per trillion).

During Tier 4 of the National Dioxin Survey, EPA examined fly ash samples from 74 sites around the US representing 22 combustion source categories. Among those sites were three utility boilers, although the fuel used in these boilers was not specified. The purpose of the sampling and analyses was to test the hypothesis that levels of PCDD/PCDFs in ash samples could be used as a surrogate measure of the concentrations of these compounds in the flue gases. The study ultimately determined that fly ash measurements were not a reliable surrogate for flue gas concentrations. Kuykendal *et al.* summarized the data from the ash sampling program in a 1989 article. The data for three different ash types (economizer ash, baghouse ash, and electrostatic precipitator ash) derived from utility boilers indicate that only OCDD was found in all three samples, at low part per trillion levels (10-70 ppt). Two samples contained unspecified hexachloro- and/or heptachlorodibenzofuran (HxCDF, HpCDF) isomers at similar concentrations (10-200 ppt). As with the data from Czuczwa and Hites, when the 1989 TEF values are applied to these results, the levels decrease by one to three orders of magnitude.

Crummett, at Dow Chemical, has published a number of articles with various co-authors on the "Trace Chemistries of Fire." The central thrust of these articles is that the apparent ubiquitous distribution of the chlorinated dioxins and furans is due to their production during many combustion processes. These processes include most all uses of fossil fuels as well as wood. In a 1984 article by Crummett and Townsend, the authors cite a number of studies of coal combustion products. In addition to those studies cited above, they cite negative findings by Harless and Lewis (1980), De Roos and Bjorseth (1979), and Redford *et al.* Crummett and Townsend cite the positive findings of Chiu *et al.* (1983) and Czuczwa and Hites (1984). They also cite the work of Eklund and Stromberg (1983), discussed above, as detecting *"PCDDs and PCDFs in two coal-fired combustors in Sweden."* However, our review of this short work indicates that the only PCDDs/PCDFs reported by Eklund

and Stromberg were from a refuse incinerator, *not* the two coal-fired facilities that were sampled. Thus, the validity of Crummett's claims about coal combustion is unclear.

Mahle and Whiting (1980) conducted laboratory experiments on the PCDDs produced from the combustion of coal. They burned coal at 600°C in the presence of air and sources of inorganic chloride (sodium chloride or hydrochloric acid) or chlorine gas, and found that TCDD, HxCDD, HpCDD, and OCDD were present in the reaction products from the two experiments involving coal, air, and either hydrochloric acid or chlorine gas. They found levels ranging from 1.2 ppb for TCDD to 64 ppb for OCDD. No PCDDs were detected in the uncombusted coal. Because coal is known to contain inorganic chlorides, often at levels of 500-1000 ppm, Mahle and Whiting performed an experiment in which sodium chloride was mixed with the coal and combusted. HpCDD and OCDD isomers were detected at low levels (0.5 and 2.7 ppb, respectively) in the combustion products of the coal/sodium chloride mixture. However, these levels were similar to the levels (0.6 and 1.3 ppb) found from the combustion of coal and air alone, leading the authors to conclude that the results did *"not indicate real differences in chlorodioxin levels when compared to the coal with air experiment."* These isomers also predominated in the experiments where TCDD was detected.

Mahle and Whiting trapped volatile combustion products and analyzed them separately from the solid residues that remained in the combustion tube. While the volatile combustion products may be somewhat similar to flue gases, and the solid residue would encompass some forms of ash, the authors acknowledge that their experiments *"were not meant as a direct simulation of a coal powerhouse or any other industrial process."* The chloride or chlorine sources were added in great excess over that which might be present in a coal-fired boiler, specifically to favor the formation of PCDDs. Similarly, the combustion temperature of 600°C was chosen as high enough to favor PCDD formation, but not so high as to result in thermal destruction of these compounds. Therefore, the findings of Mahle and Whiting must be viewed relative to the typical combustion conditions and products present in a coal-fired powerhouse.

Phase 2 Findings

The second phase of the review of coal combustion uncovered a number of articles or reports published in late 1992 and later that addressed dioxins and furans in relation to coal combustion. The

most significant documents included a comprehensive review entitled "Organic Compounds from Coal Utilisation," by Sloss and Smith, published by the International Energy Agency (IEA) Coal Research group in October 1993. The second major document was a report dated October 1994 prepared by the American Electric Power Service Company for the U.S. Department of Energy on "A Study of Hazardous Air Pollutants at the Tidd PFBC Demonstration Plant." Five other relevant papers were identified, three published in *Chemosphere.*

Harrad, Fernandes, and Creaser (1991) investigated domestic coal combustion as a possible source of PCDDs/PCDFs found in soils in Great Britain. They analyzed samples of soot collected from domestic chimneys where coal was used a fireplace fuel. They also examined fly ash from a municipal waste incinerator. All analyses were conducted using a low resolution GC/MS instrument. They reported concentrations of 2,3,7,8-TCDD in soot ranging from 3 to 1200 ng/kg and similar concentrations of 1,2,3,7,8-pentachlorodibenzo-*p*-dioxin (PeCDD). "Total" concentrations were reported as well, generally at higher levels than those cited above. Based on the brief description of the methods used, the 2,3,7,8-TCDD levels reported are mostly below demonstrated calibration range of the instrument used and undoubtedly have large associated uncertainties. In addition, the soot samples analyzed do not represent residues that would be found in a coal-fired boiler or power generating plant. The soot samples come from open draft fireplaces burning coal as a fuel. The relationship between the combustion that takes place in such a fireplace is even more remote from a coal-fired boiler than a wood stove is from an open fire.

Work by Weinecke, Kruse, and Wasserman in 1992 involved organic compounds identified in flue gases from a coal-fired power station in Flensburg, Germany. The authors examined the chloride content of four different coal species available to the power plant and attempted to correlate the chloride content of the coal with the emissions of halogenated organics in the flue gases. They employed a low resolution GC/MS instrument and make a statement that *"Polychlorinated dibenzodioxins or dibenzofurans are below the detection limit (1 ng/Nm^3).*" They did not examine any ash matrices. In addition, the use of a low resolution instrument virtually precludes any meaningful measurement of PCDDs/PCDFs in either the flue gases or any other matrices, at the levels of interest for these compounds.

Gohda *et al.* (1993) reported on the analyses of fly ash from coal-burning steam engine and a coal gasification plant in China. According to this report, bottom ash from the steam engine contained 140 ng/kg of total PCDD and 870 ng/kg of total PCDF, comprised of the hepta- and octachlorinated congeners. No data were presented for any individual congeners. Coal fly ash samples were collected at the cyclone, bag house, and exhaust of the coal gasification plant and analyzed. The authors report that *"surprisingly low levels of PCDF/PCDF were detected"* but they do not provide even a "total" concentration in support of this assessment. They report on an attempt to determine a reason why so little PCDDs/PCDFs were observed. The authors used ^{13}C-labeled pentachlorophenol as a precursor for dioxins. They added this precursor, which would never be found naturally with all the carbons containing ^{13}C, to samples of coal ash, heated the ash samples, and measured the PCDDs/PCDFs that were formed. Experiments were conducted using the coal fly ash as collected and following an extensive extraction of the ash using toluene. Ash that was extracted did produce "surprisingly high levels" of ^{13}C-labeled PCDDs, while the untreated ash did not. Although no specific numerical results are presented, the authors state that the amounts of PCDDs/PCDFs produced by the treated coal ash are comparable to the levels found in municipal solid waste incinerator ash. However, the untreated ash apparently produced concentrations *"1 to 2 %"* of that found in incinerator ash. Unfortunately, the details of the analytical methods are skeletal, at best, although it is clear that a low resolution GC/MS instrument was employed, severely limiting the sensitivity.

Addink and Olie (1995) wrote a critical review article on the mechanisms of formation and destruction of PCDDs and PCDFs. This paper examines the state of knowledge regarding these mechanisms and provides information on the processes that influence the formation of PCDDs/PCDFs through *de novo* synthesis. The paper includes a lengthy discussion of the reaction mechanisms that may lead to PCDD/PCDF formation. The authors did not measure PCDDs/PCDFs in matrices collected from combustion facilities, but did use fly ash from an unspecified source as one of many "reactants" in a series of laboratory experiments. Throughout this work, the authors distinguish between "collected" fly ash, such as that trapped by a pollution control device like an electrostatic precipitator, and "uncollected" fly ash, the material that is emitted into the environment with the flue gas. Among their findings are the observations that *"De novo synthesis from carbon appears to be a long time scale process (hours) and takes place on collected fly ash particles only."* They reason

that fly ash that escapes the facility does not have sufficient residence time at the temperatures required for *de novo* synthesis of PCDDs/PCDFs. While the mechanisms described by the authors might be relevant to processes occurring during coal combustion, it is important to note that this paper does not address coal combustion or coah ash directly.

Grochowalski and Wybraniec (1996) investigated the levels of PCDDs/PCDFs in flue gases and fly ash from a coal-fired power plant in Poland. They report congener-specific results for the seventeen 2,3,7,8-substituted PCDDs/PCDFs in flue gases and fly ash. Because of interest in burning hazardous wastes in various combustion facilities, the authors investigated the flue gases and fly ash produced under three sets of combustion conditions: coal burning alone, coal burned in the presence of a gaseous organochlorine source (vinyl chloride), and coal burned in the presence of a liquid organochlorine source (e.g., chlorinated solvents).

The report is troubling in several respects. First, there is a significant typographical error in the table for the fly ash results. The table header is identical to that for the flue gas results, stating that the results are for flue gas and using gas units for the concentrations. However, units appropriate for ash samples are given at the bottom of the same table. While one would hope that this is a simple error in typesetting the article, it does cast some doubt on the actual results reported in the table. Secondly, in describing the analytical methodology, the report refers to two EPA methods for PCDDS/PCDFs, Methods 1613 and 8280. Method 1613 is a high resolution GC/MS method employing very specialized instrumentation and Method 8280 is a low resolution method employing a much more commonly available GC/MS instrument. From the description in the paper, it is obvious that the authors employed a low resolution GC/MS instrument. The mention and apparent use of techniques from a high resolution method is troubling, in that they have little relevance to the low resolution instrumentation.

Most troubling of all are some of the actual results presented. For example, for the flue gas samples, the authors report a concentration of 2,3,7,8-TCDD of 0.002 ng Nm^{-3} for both the coal combusted alone and the coal plus the liquid waste containing organochlorine. For the coal combusted with vinyl chloride, the level is 0.001 ng Nm^{-3}, despite the fact that vinyl chloride is a ready source of chlorine that would be expected to produce PCDDs. Both of these values are

probably below the demonstrated capability of the instrumentation and method that were employed. Moreover, they suggest that the background levels in the sampling equipment, laboratory, instrumentation, or some combination of the three, are what drives these measurements. The flue gas levels found for other congeners show similar patterns. The report contains no mention of either collecting or analyzing blanks.

Assuming that units for the fly ash are a typographical error in the table, the specific results still are cause for concern. The reported levels of PCDDs/PCDFs are roughly 500 times lower than the accepted performance of a low resolution GC/MS method. Even allowing for an extrapolation of results below that range, these values strain the limits of credulity, particularly in light of the concerns about background levels mentioned above. Thus, the reviewer believes that the results of this study should be viewed with extreme skepticism.

The IEA Coal Research report by Sloss and Smith (1993) provides a thorough and comprehensive review of the organic emissions resulting form the use of coal. It includes chapters on the compounds of environmental interest (including PCDDs and PCDFs), the chemistry of formation of these compounds, emissions from coal combustion, emissions from non-combustion sources, environmental pathways and concentrations, and environmental effects.

This report describes various reaction mechanisms that may lead to the formation of PCDDs/PCDFs and other organic compounds of concern. It provides a detailed discussion of the effects of combustion conditions and conditions downstream from the combustion zone itself. In particular, the authors examined in detail the untested suggestion by earlier researchers that the reactions that lead to PCDD/PCDF formation in waste incineration systems may also occur in coal combustion systems.

The authors note that the flue gases in a coal-fired power plant cool rapidly after exiting the combustion zone. They state that electrostatic precipitators (ESPs) are *"normally operated at around 150°C, so that it unlikely that PCDD/PCDF would be produced there."* However, they go on to note that *"hot-side ESP systems operate at around 350°C,"* a temperature within the range for possible *de novo* synthesis. They also note that the residence time of possible precursor molecules

for PCDDs/PCDFs *"in the ESP of coal-fired power plants are similar to those in incineration systems."*

The authors discuss the role of fly ash particles as a catalytic surface on which PCDDs/PCDFs may form. They discuss work by various investigators that supports a mechanism whereby chlorine, metals, and metal compounds in the fly ash of waste incinerators may result in the formation of PCDDs/PCDFs. They conclude this section of the report with a statement that these conditions *"are not provided to any great extent by fly ash from coal-fired power plants."* The report continues by pointing out that sulfur dioxide (SO_2) inhibits the formation of PCDDs/PCDFs and is more common in coal-fired flue gases than in waste incinerators.

The overall conclusion of the formation chemistry chapter is that coal-fired power plants differ from waste incinerators in the temperature of the ESP device, which is why PCDDs/PCDFs are not likely to be found in coal ash.

The chapter in the report on emissions from combustion sources summarizes data from other researchers with regard to the emissions of PCDDs/PCDFs from various sources. Of specific interest are results from Davies *et al.* (1992) and Williamson (1993) regarding TCDD and TCDF emissions in flue gases. From the summary of these results, it appears that low levels of 2,3,7,8-TCDF and some HxCDDs were emitted from a coal-fired power station designed to generate 380 MWe. Unfortunately, the report by Davies *et al.* (1992) was not available to the reviewers and the citation for Williamson (1993) is listed as "private communication."

The general consensus of the Sloss and Smith report is that PCDDs/PCDFs *may* be produced by coal combustion processes. As noted by the authors at the end of the chapter on combustion emissions, the technologies designed to limit the flue gas emissions of PCDDs/PCDFs may concentrate them in the liquid or solid wastes from the control devices.

The conclusions of Sloss and Smith were substantiated by later work by Brain *et al.* (1995), who examined "toxic organic compounds" from coal combustion. These toxic organics included PCDDs/PCDFs, polychlorinated biphenyls (PCBs), and the brominated analogs of the

PCDDs/PCDFS, the PBDDs/PBDFs. These authors sampled the flue gases from 13 different combustion sources, including open grate domestic coal-burning appliances, nine types of coal-fired industrial boilers from 13 kW to 43 MW capacity, one pulverized fuel burning cement kiln, an oil-fired boiler, and a waste incinerator.

Brain *et al.* concluded that the highest dioxin emissions, measured as toxic equivalents of TCDD (i.e., the TCDD TEQ approach), were from coal-fired domestic appliances. Their measurements of the emissions from coal-fired power generating stations were substantially below those from the domestic appliances. The authors noted that all of their measured emission rates for coal-fired boilers were below even the most stringent European air quality standard of 0.1 ng Nm^{-3}. Only the domestic appliances burning bituminous coal in an open fire would exceed that European standard.

Unfortunately, Brain *et al.* did not investigate any ash matrices. The nine coal-fired boiler employed a variety of pollution control approaches, ranging from none to an electrostatic precipitator on the 43 MW circulating fluidized bed boiler.

The report on the pressurized fluidized bed combustor (PFBC) demonstration study conducted by the American Electric Power Service Company of Columbus, Ohio, supports many of these same conclusions. The report describes a test of a hot gas clean up system for advanced particle filtration (APF). The system contains a series of ceramic cyhinders, called "candles," to filter the flue gases exiting a cyclone system. The hot gases pass from the outside of the ceramic candle to the inside and exit the filtration device. The candles remove particles from the flue gases and the particles are removed from the filtration device for disposal. In the demonstration study, the APF unit was installed after the primary cyclone system. The gases exiting one of the seven primary cyclones was diverted through the APF and then recombined with the other gases after the secondary cyclones and directed through the electrostatic precipitator unit. It processed about one-seventh of the total gas flow through the system.

During the demonstration study, PCDD/PCDF samples were collected from four sources of flue gases and three types of ash. Ash was collected from the ESP, the primary cyclone, and the APF

unit. No PCDDs/PCDFs were detected in the cyclone ash. One specific isomer, 2,3,4,6,7,8-HxCDF, was reported in the APF ash at an average concentration of 0.3 ng/kg, across all the runs, and 1,2,3,4,6,7,8-HpCDD was reported in the ESP ash at an average concentration of 1.4 ng/kg. Results for the ESP ash also included results for several congener totals, including Total TCDD, Total PeCDD, Total HxCDD, and Total HpCDD. Neither 2,3,7,8-TCDD nor 2,3,7,8-TCDF were found in the APF ash, and 2,3,7,8-TCDF was detected in only one of the ESP ash samples.

The demonstration study involved tests of several different sets of operating conditions over a number of different "runs." In reviewing the report, SAIC has used the results from Run 3 as an example. The table below provides data from the PFBC report for Run 3, focusing on the ESP unit. While this table makes no attempt to determine a mass balance, it does provide some indication of the possible transfer of PCDDs/PCDFs from the flue gases to the ESP ash. Any analytes not listed in the table were not reported as detected at any of the three sampling points. The data qualifiers listed in the table and described below were taken from the demonstration plant report.

Analyte	Run 3 Concentration at ESP Inlet (pg/Nm^3)	Run 3 Concentration at ESP Outlet (pg/Nm^3)	Run 3 Concentration in ESP Ash (ng/kg)
2,3,7,8,-TCDF	1.7 E	ND	0.21
1,2,3,4,7,8,-HxCDF	1.6 E	1.5 EB	ND
2,3,4,6,7,8-HxCDF	3.3 EB	3.9 B	ND
1,2,3,4,6,7,8-HpCDD	3.3 E	11	1.8
1,2,3,4,6,7,8-HpCDF	1.6 E	7.3 E	ND
OCDD	34	110	1.6
OCDF	ND	20	ND

ND = not detected
E = Value is an estimated upper limit
B = Analyte also found in the associated method blank at ≥30% of sample result

The results for 1,2,3,4,7,8-HxCDF and 2,3,4,6,7,8-HxCDF in Run 3 are discounted by SAIC because of concerns about the levels found in the associated method blanks. The level of 1,2,3,4,6,7,8-HxCDF in the blank associated with the gas samples was 1.4 pg/Nm^3, and the inlet and outlet results of 1.6 and 1.5 cannot be distinguished from that amount. The blank also contained 2.1

pg/Nm^3 of 2,3,4,6,7,8-HxCDF, which accounts for more than half of the amount found in either the inlet or outlet sample.

Many of the other results are reported as estimated upper limits, suggesting that there were either interferences which precluded truly quantitative measurement of the analyte or the results exceeded the upper limit of the calibration range of the instrumentation. The estimated nature of these results makes it difficult to draw firm conclusions. However, the data for 2,3,7,8,-TCDF suggest that this analyte is effectively removed from the flue gases in the ESP unit, resulting in the detection of this analyte in the ESP ash. The results for 1,2,3,4,6,7,8-HpCDD, 1,2,3,4,6,7,8-HpCDF, OCDD, and OCDF suggest that these analytes may be formed in the ESP unit, since the outlet concentrations are higher than those at the inlet. Two of these analytes also appear in the ash, suggesting at least partial removal in the ESP unit.

The data from Run 3 were chosen for this example because this was the only one of the runs in which 2,3,7,8-TCDF was reported. The other analytes generally showed similar trends in the other runs. SAIC has not evaluated the specific run conditions, nor examined whether the conditions in Run 3 were likely to be employed during "normal" plant operations. Unfortunately, data on the temperature of the ESP device could not be readily located in the report by the reviewer and thus, it could not be compared to the conclusions of the report by Sloss and Smith, that coal-fired power plant ESP devices are operated at lower temperatures than those on waste incinerators, thereby reducing the potential for producing PCDDs/PCDFs.

Phase 3 Findings

In late March 1998, SAIC received a copy a report by the Electric Power Research Institute (EPRI) entitled *"PCDDs and PCDFs in Coal Combustion Byproducts (CCBs)."* The report represents that first study directed specifically at answering the question of whether PCDDs/PCDFs are found in ash from coal-fired power plants. At EPA's request, SAIC reviewed this report in April 1998.

According to the report, samples were collected from eleven landfills and impoundments where coal combustion byproducts and other power plant wastes are comanaged. A total of 15 samples were collected, apparently as cores taken through the landfill. The EPRI report provides no other detail on the sample collection procedures, thus no assessment can be made of the representativeness of these 15 cores to the material in each landfill or impoundment. It is also not possible to determine the specific types of ash that were sampled, e.g., bottom ash, fly ash, or both.

The EPRI report provides the summary level analytical results for the 15 samples as well as for some of the laboratory quality control samples that were associated with these core samples. The report also contains a discussion of a data validation effort that was conducted on these results. In general, the report is quite comprehensive. However, it does contain several factual errors or statements that raise concerns for the SAIC reviewers. Fortunately, the overall effects of these errors and statements are not severe, but do require some mention here.

Specifically, the second sentence of the Sample Analysis and Data Review section of the report (page 2-1) states that the analytical method used for the 15 samples (SW-846 Method 8290) is *"the only analytical method for PCDDs and PCDFs accepted by the EPA for hazardous waste characterization."* This statement is at least partially untrue. The Office of Solid Waste has stated repeatedly that "any appropriate method" may be used for the characterization of hazardous waste. In the case of PCDDs/PCDFs, OSW has accepted results from EPA Method 1613, a method published by the Office of Water, for various waste characterization purposes. The issue of what methods may be used for waste characterization is a commonly misunderstood one. The only actual restriction under the RCRA program is that a delisting petition must employ SW-846 methods in order to demonstrate that the waste is eligible for delisting. The significance of this issue affects the validation of the sample results, and is discussed in greater detail later in our report.

The second area of concern with the EPRI report is that the PCDD/PCDF data generated using Method 8290 were validated using the *"National Functional Guidelines for evaluating organic analytes."* The guidelines in question were developed by the National Contract Laboratory Program (CLP) for use in validating data generated using methods specific to that EPA program. The relationship of the guidelines to other analytical methods and other EPA programs is minimal,

especially for analyses of PCDDs/PCDFs by a high resolution mass spectrometric method such as Method 8290. SAIC believes that the likely effect of the validator's use of the CLP guidelines is to short change the quality of the ash sample results, making them appear less useful than they would be if more appropriate validation criteria were employed.

SAIC reviewed the summary level results for the 15 samples as well as the EPRI data validation discussion. The EPRI report noted a problem with two blank contaminants, 2,3,4,6,7,8-HxCDF and OCDD in one blank associated with 11 of the 15 samples. The levels of 2,3,4,6,7,8-HxCDF in the 11 samples were virtually identical to the level found in the associated method blank. Thus, there is no evidence that this HxCDF was present in the 11 samples. In contrast, OCDD was found in eight samples at levels that were 6 to 50 times greater than the amount found in the associated method blank. Low ng/kg levels of OCDD are not uncommon in laboratory method blanks. Given the fact that the six of the eight sample results were at least 10 times higher than the blank level, there is no reason to qualify those six results. The results for the other two samples in which OCDD was identified were 6-8 times the blank level, suggesting that there may be some uncertainty in the actual sample results, but clearly indicating that the analyte was present in the samples.

The EPRI validation report describes problems with internal standards and surrogates that are "under-recovered" and "over-recovered" in some samples. Unfortunately, the EPRI report does not provide the recovery data for these isotopically labeled standards, so the potential problems could not be thoroughly evaluated by SAIC. The existing acceptance criteria in Method 8290 are recoveries of 40-130% for the labeled analytes. These limits are simple consensus limits, developed in about 1989 when the method was first drafted. Those limits have never be revised by EPA, nor has the method undergone a formal interlaboratory validation study. In contrast, EPA Method 1613, mentioned earlier in this review, was originally published in 1991 with consensus limits of 25-150%. Those limits were subsequently revised based on an international interlaboratory study and widened somewhat for a few analytes. Method 1613 is very similar to Method 8290, with the one major exception that it employs 15 labeled analytes as "internal standards," allowing isotope dilution quantitation to be employed for 15 of the 17 target analytes in the method. In contrast, Method 8290

employs only 10 labeled analytes for quantitation, using the other five as "surrogates," thereby losing the benefit of true isotope dilution and an accurate recovery correction for those five analytes.

Given the wider QC limits in Method 1613, it is quite possible that the analytes for which the EPRI report noted recovery problems would have had acceptable recoveries, thereby minimizing the number of results that would be qualified by the reviewers. Unfortunately, that supposition cannot be tested without the recovery data.

The EPRI report goes to great lengths to evaluate a number of data reporting scenarios with regard to the toxic equivalent concentration (TEQ) of 2,3,7,8-TCDD. In particular, the EPRI report describes four scenarios for calculating the TEQ value for each sample with non-detected analytes and estimated values:

1. Using the reported detection limits as if they were detected concentrations
2 . Using one-half the reported detection limits as the concentrations
3. Non-detects set to zero and estimated maximum possible concentrations (EMPCs) as actual concentrations
4. Non-detects set to zero and EMPCs set to one-half their estimated concentration

As noted in the EPRI report, the use of the first scenario is the "most conservative" with respect to protecting the environment, in that it greatly overstates the likely risk posed by the sample. In fact, the laboratory method blanks have a higher TEQ than some of the samples. The second scenario, while commonly used in some EPA programs, may still overestimate the TEQ of the sample.

The EMPC values represent target analytes that do not meet one specific aspect of the method-specified identified criteria, the ratio of the abundances of the two quantitation ions monitored for each analyte. The fact that the ion abundance ratio criterion is not met suggests that there is some other component in the sample that created a positive interference for one of the quantitation ions. That same interference would be expected to raise the calculated concentration as well. Thus, the result is termed an estimate of maximum possible concentration.

The EPRI data contain a total of 10 EMPC values. SAIC has reviewed these results and compared them to the reported results for the method blanks and the detection limits calculated for each analyte in each sample. Six of the 10 EMPC values are associated with analytes which also appeared in the associated method blank, the 2,3,4,6,7,8-HxCDF and OCDD results discussed above. All six of those EMPC values would have been rejected on the basis of the blank results, even if the result had met all of the identification criteria. The remaining four EMPC value are all at concentrations below those that are reporped as sample-specific detection limits for the same analytes in other samples. Three of those four EMPCs were for 2,3,7,8-TCDF, yet were reported at concentrations below the detection limits for the majority of the other samples (e.g., 0.1 to 0.2 ng/kg).

In reviewing the occurrence of 2,3,7,8-TCDF in other samples, SAIC notes that this second-most toxic analyte was unambiguously identified in three of the 15 coal ash samples. It was also an EMPC is three other samples. The three EMPC values are at concentrations below those reported for the three unambiguous identifications of 2,3,7,8-TCDF. In two of the three unambiguous identifications, the sample also had a measurable concentration of 1,2,3,4,6,7,8-HpCDD. Two of the EMPC values for 2,3,7,8-TCDF were also associated with measurable concentration of this HpCDD. Thus, based on the similarity of the pattern of PCDDs/PCDFs in this small data set, SAIC believes that it is reasonable to accept the three EMPC values for 2,3,7,8-TCDF as actual concentrations in any further evaluations such as the TEQ. SAIC believes that the fourth EMPC value, for 2,3,4,6,7,8-HxCDF in one sample, can be discounted and not used in further evaluations.

Based on these considerations, SAIC has used the TEQ results provided in Table A-4 of the EPRI report. The TEQ values for the 15 ash samples range from 0.0 to 0.064 ng/kg. This is equivalent to the range of 0 to 64 pg/kg TEQ, or parts per quadrillion. These TEQ values are exceedingly low. The TEQ values are almost 100 times lower than the level at which EPA has granted an exemption from hazardous waste regulation. Moreover, the levels of PCDDs/PCDFs found in these coal ash samples are four to five orders of magnitude lower than some of the early literature values (e.g., Chiu *et al.* , 1983, and Czuczwa and Hites, 1984).

A total of six of the 15 samples contained detectable, but very low, concentrations of 2,3,7,8-TCDF. The highest of the actual 2,3,7,8-TCDF values for any of the samples was 0.41 ng/kg. The highest level of OCDD in any of the samples was 22.6 ng/kg. No 2,3,7,8-TCDD was found in any of the 15 coal ash samples.

In summary, the EPRI data set demonstrates that while PCDDs/PCDFs can be detected in ash from coal combustion, the levels are very low compared to other combustion sources such as municipal waste combustion. The patterns of the few PCDDs/PCDFs that were found are predominated by the higher, less toxic, analytes such as 1,2,3,4,6,7,8-HpCDD and OCDD.

CONCLUSIONS

Compared to municipal waste incineration, the products of coal combustion have received little attention until relatively recently. Based on our review of the literature, it appears that some PCDD/PCDF isomers may be produced during coal combustion in industrial settings, e.g., power plants. The theoretical groundwork that describes the formation of these contaminants has been established by several sources. The work of Sloss and Smith (1993) provides a firm foundation for future investigations.

The strength of the principal conclusion of Sloss and Smith, that PCDDs/PCDFs are not likely to be formed in the electrostatic precipitators of coal-fired power plants because of the temperature at which such units operate, is weakened by the findings from the PFBC pilot plant study. However, the reviewers were not able to confirm the temperature at which the ESP operated during that study, and therefore, the conclusions of Sloss and Smith may hold true, provided that ESP temperatures are considered.

Only one analyte, 1,2,3,4,6,7,8-HpCDD, was observed in the ESP ash consistently in Runs 1-3 at the PFBC demonstration plant. This suggests that the conditions in Run 3 favored the formation and collection of the other PCDDs/PCDFs reported for that run.

The PFBC pilot plant differs from other coal-fired boilers in the use of the advanced filtration device, but the effects of this unit could not be fully evaluated, since it receives only a portion of the flue gas stream leaving the primary cyclones. The same analyte reported in the ESP ash, 1,2,3,4,6,7,8-HpCDD, was also the only analyte reported in the APF ash.

It is important to note that 2,3,7,8-TCDD was not reported in ash in the PFBC study(and 2,3,7,8-TCDF was only reported in one of the three runs. The concentrations of the other isomers that were found in ash samples were at the lower end of the capabilities of the analytical methods and instrumentation. When multiplied by the internationally accepted toxicity equivalency factors (TEFs), the resulting TCDD-equivalent concentration would be below the level for which EPA has previously granted an exemption from hazardous waste regulation.

The report of 2,3,7,8-TCDD in coal soot by Harrad *et al.* (1991) is important in the context of the purpose of that study - namely to identify historical sources of the PCDD/PCDF concentrations found in soils in Great Britain. However, the relevance of those findings to the combustion of coal in power plants and other industrial settings is seriously in doubt.

The work by Gohda *et al.* (1993) supports the theoretical work of Sloss and Smith, suggesting that PCDDs/PCDFs may be produced during coal combustion. However, the specifics of that report are so vague that it is difficult to determine either the isomers that were identified or their apparent levels.

The result presented by Grochowalski and Wybraniec (1996) are potentially seriously flawed. The concerns include obvious errors in the written publication as well as serious methodological issues and an internally inconsistent data set.

The results of Davies *et al.* and Williamson, cited in Sloss and Smith, could not be corroborated by the reviewers. However, those results suggest that low levels of 2,3,7,8-TCDF may be present in *flue gases*, not ash.

The EPRI study provides clear evidence that a small number of PCDDs/PCDFs may be produced during coal combustion and are present in ash samples. However, the levels are very low and not consistently found in all samples. No 2,3,7,8-TCDD was found in any of the 15 ash samples in the EPRI study and 2,3,7,8-TCDF was only found in six of the 15 samples.

In summary, the EPRI study demonstrates that there is formation of very low levels of some PCDDs/PCDFs during coal combustion. The data from the PFBC study suggest that some PCDDs/PCDFs may be formed in the flue gases during passage through the ESP unit at that facility. The presence of PCDDs/PCDFs in the ESP ash appears to be related to the run conditions, as there were marked differences amongst the results for the various runs. The relationship of the ESP operating temperature and PCDD/PCDF production may bear further examination.

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APPENDIX O

ARSENIC AND BERYLLIUM IN FFC WASTES

APPENDIX O. ARSENIC AND BERYLLIUM IN FFC WASTES

Arsenic and beryllium consistently exceeded target risk levels in high-end deterministic FFC waste modeling scenarios. They both display low health-based levels (HBLs) based on a cancer slope factor (CSF); the HBLs calculated by this method are well below analytical detection limits. Additionally, data for determining the cancer slope factor for beryllium has recently (April 1998) been determined to be inadequate. However, other levels for comparison are available for these two constituents: an oral reference dose (RfD)-based health-based level and a maximum contaminant level (MCL). This appendix discusses the bases for all three benchmarks; compares the receptor well concentrations to the CSF-based HBL, RfD-based HBL, and MCL; and discusses how MCLs and HBLs have been used for decision making in recent rulemakings.

BASIS FOR CSF, RFD, AND MCL

Arsenic and beryllium are both listed on EPA's Integrated Risk Information System (IRIS, http://www.epa.gov/iris/). Both carcinogenic and noncarcinogenic effects are listed, as well as corresponding CSFs and RfDs. The last entry in IRIS for either of these constituents is April 1998, which is after the publication of EPA's newest cancer guidelines (April 23, 1996). However, some of the nomenclature paraphrased from IRIS and presented in this appendix may be inconsistent with the proposed guidelines.

Arsenic is categorized as an A (human) carcinogen, based on "sufficient evidence from human data." Studies have noted increased cancers of various organs in populations consuming drinking water high in inorganic arsenic and increased lung cancer mortality from human populations exposed primarily through inhalation. In regard to noncarcinogenic effects, IRIS considers confidence in the RfD to be "medium," based on data characterizing oral exposure to human populations. The MCL for arsenic is 0.05 mg/L, as listed in 40 CFR Section 141.11. This value, which was used as a drinking water standard as far back as 1943, was established as an MCL in 1975. EPA noted long-term chronic effects at 300 to 2,750 µg/L, but no illness at 120 µg/L ("Arsenic in Drinking Water: Regulatory History," EPA Office of Ground Water and Drinking Water, http://www.epa.gov/ogwdw000/ars/ars1.html).

Beryllium is categorized as a B1 (probable human) carcinogen. Specifically, inhaled beryllium would be categorized as a "likely" carcinogen in humans, and the human carcinogenic potential of ingested beryllium cannot be determined. In regard to noncarcinogenic effects, the oral RfD was recently recalculated from 0.005 to 0.002 mg/kg-day. IRIS considers confidence in the RfD to be "low to medium." The primary MCL for beryllium of 0.004 mg/L represents an MCLG, or MCL goal. Many carcinogens have MCLGs equal to zero, but for beryllium and other carcinogens with limited data, the MCLG is calculated from the RfD with an additional safety factor to account for cancer risk (57 FR 31784, July 17, 1992).

COMPARISON OF RECEPTOR WELL CONCENTRATIONS TO HBLS AND MCLS

FFC waste characterization data availability differs for arsenic and beryllium. In general, arsenic is better characterized than beryllium (see Table O-1). As shown in Table O-2, no data are available to characterize leachable beryllium levels in oil combustion wastes because no TCLP or EP beryllium analyses were presented in EPRI's oil ash database.

USE OF MCLS IN RECENT OSW RULEMAKINGS

Two petroleum refining wastes were proposed for listing as hazardous (November 20, 1995) based, in part, on the leachability of arsenic. EPA determined that arsenic was one of the primary constituents of concern, based on the ingestion of groundwater contaminated from waste disposed in a landfill. EPA used the above referenced CSF (from IRIS) in its risk assessment and did not consider arsenic's MCL. No public comments were received arguing that arsenic's MCL should have been considered. This rule is scheduled to be finalized summer 1998.

The proposed Hazardous Waste Identification Rule (HWIR, December 21, 1995) identified exit levels for hazardous wastes. The rule proposed that if a hazardous waste, when analyzed, has leachate and total constituent levels below those proposed it would no longer be regulated as hazardous. Two sets of exit levels were proposed: one set based on MCLs and another based solely on RfDs and CSFs.

Table O-1. Arsenic in FFC Wastes

Scenario	95th % Observed Conc. (mg/L)	DAF Result from Modeling	High-end Risk or HQ for Adult Resident Based on:		
			CSF-based HBL (0.00029 mg/L)	RfD-based HBL (0.015 mg/L)	MCL (0.05 mg/L)
Coal-fired Utility Surface Impoundment (Scenario CS)	9.64	65.4	***5.08×10^{-4}***	***9.83***	***2.95***
Coal-fired Utility Landfill (Scenario CL)	9.64	3.17	***1.05×10^{-2}***	***203***	***60.8***
Coal-fired Utility Minefill (Scenario CF)	9.64	3.90	***8.52×10^{-3}***	***165***	***49.4***
Oil-fired Utility Surface Impoundment (Scenario OS)	4.15	56.0	***2.56×10^{-4}***	***4.94***	***1.48***
Oil-fired Utility Monofill (Scenario OM)	4.15	158	***9.06×10^{-5}***	***1.75***	0.53
Oil-fired Utility Subtitle D Landfill (Scenario OL)	4.15	6.04	***2.37×10^{-3}***	***45.8***	***13.7***
FBC Landfill (Scenario FL)	0.35	2.15	***5.61×10^{-4}***	***10.9***	***3.26***
FBC Minefill (Scenario FF)	0.35	3.58	***3.37×10^{-4}***	***6.52***	***1.96***
Non-utility Monofill (Scenario NM)	9.64	4.61	***7.21×10^{-3}***	***139***	***41.8***
Non-utility Subtitle D Landfill (Scenario NL)	9.64	2.80	***1.19×10^{-2}***	***230***	***68.9***

Note: Values in bold italics exceed the risk threshold (HQ=1 or risk=10^{-6}).

The ground-water pathway risk assessment for cement kiln dust used both MCLs and RfD or CSF-derived HBLs ("Technical Background Document: Human Health and Environmental Risk Assessment in Support of the Report to Congress on Cement Kiln Dust Waste," December 1993). Specifically, concentrations of arsenic and beryllium were compared to multiples of the MCL and the CSF-generated HBL in assessing risks from ground water and surface water in the screening analysis.

Table O-2. Beryllium in FFC Wastes

Scenario	95th % Observed Conc. (mg/L)	DAF Result from Modeling	High-end Risk or HQ for Adult Resident Based on:		
			CSF-based HBL (Inadequate)	RfD-based HBL (0.10 mg/L)	MCL (0.004 mg/L)
Coal-fired Utility Surface Impoundment (Scenario CS)	Insufficient data		Inadequate	---	---
Coal-fired Utility Landfill (Scenario CL)	Insufficient data		Inadequate	---	---
Coal-fired Utility Minefill (Scenario CF)	Insufficient data		Inadequate	---	---
Oil-fired Utility Surface Impoundment (Scenario OS)	Insufficient data		Inadequate	---	---
Oil-fired Utility Monofill (Scenario OM)	Insufficient data		Inadequate	---	---
Oil-fired Utility Subtitle D Landfill (Scenario OL)	Insufficient data		Inadequate	---	---
FBC Landfill (Scenario FL)	0.28	2.04	Inadequate	***1.37***	***34.3***
FBC Minefill (Scenario FF)	0.28	2.2	Inadequate	***1.28***	***31.8***
Non-utility Monofill (Scenario NM)	Insufficient data		Inadequate	---	---
Non-utility Subtitle D Landfill (Scenario NL)	Insufficient data		Inadequate	---	---

Note: RfD of 0.002 mg/kg/day used in calculations (IRIS, April 1998). HBL derived using cancer slope factor is not presented because the oral database is considered inadequate for the assessment of carcinogenicity (IRIS, April 1998). Values in bold italics exceed the risk threshold (HQ=1).